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CROP AND FOREST LOSSES DUE TO CURRENT AND PROJECTED EMISSIONS FROM COAL-FIRED POWER PLANTS IN THE OHIO RIVER BASIN

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This report was prepared as part of the Ohio River Basin Energy Study (ORBES), a multi-disciplinary research program supported by the Environmental Protection Agency. The major objectives of this part of the ORBES research are to summarize and evaluate the literature on metabolic responses of plants to low levels of SO_x, their acidic conversion products, and oxidants, particularly O_3; assemble data on emission concentrations around representative primary emission sources in the study region; determine the distribution and acreages of sensitive crop and forest species in the areas near emission sources; review and apply damage functions for key sensitive species to estimate losses in yield from a range of current and projected pollutant levels; and estimate probable maximum and minimum effects of the emissions and transformation products on the regional yield of crop and forest products. Complete descriptions of each part of the research are contained in this report.

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FOREWORD

The Ohio River Basin Energy Study (ORBES) is an interdisciplinary, multi-university inquiry into the environmental, economic, and social impacts of energy facility development in a major portion of the Ohio River Basin. The study was initiated by the U.S. Environmental Protection Agency in 1976 in response to a directive from the U.S. Senate Appropriations Committee.

Most of the ORBES research was conducted by a group of faculty members at the University of Illinois, Indiana University, the University of Kentucky, the University of Louisville, The Ohio State University, the University of Pittsburgh, Purdue University, and West Virginia University. In a few instances, the necessary expertise was found at other universities or independent research organizations.

This report is one of a series produced in support of the overall ORBES assessment. For a summary of the entire study, see Ohio River Basin Energy Study (ORBES): Main Report, by the ORBES Core Team, also being published by the U.S. Environmental Protection Agency.

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Orie L. Loucks
Principal Investigator

CONTENTS

PAGE

FOREWORD iii

ACKNOWLEDGEMENTS iv

TABLES ix

FIGURES xii

SECTION I - INTRODUCTION 1

SECTION II - SUMMARY AND FINDINGS 5

SECTION III - A REVIEW OF MECHANISMS OF ACTION OF OZONE AND SULFUR DIOXIDE POLLUTANTS ON CROPS AND FORESTS

INTRODUCTION 19
UPTAKE OF SO_2 and O_3 BY PLANTS 19
EFFECTS OF OZONE ON CROPS AND FORESTS 21
EFFECTS OF SULFUR DIOXIDE ON CROPS AND FORESTS 28
    Biochemical Effects 28
    Physiological Disturbances 36
    Regional Implications 37

CONCLUSIONS 38
REFERENCES 39

SECTION IV - ASSESSMENT OF AIR POLLUTION INJURY TO EASTERN WHITE PINE IN INDIANA

INTRODUCTION 43
    Background Data: Ozone 43
    Sulfur Dioxide 45
    Minor Pollutants and Agents that Cause Similar Symptoms 46

METHODS 47
    Index of Injury 47
    Statistical Tests 52
    Pollutant Concentrations 52

SITE DESCRIPTIONS 53
    Bloomington Site 53
    Alamo Site 53
    Breed #1 53
    Petersburg 53

Clifty Creek 54
Breed #2 54
Lizton 54
Banta Road 54
Carmel 55
Wisconsin Sites 55
Frequency Distribution Analysis 60
Flecking/Chlorotic Mottling 60
Percent Necrosis of Needles 63
Length of Necrosis 63
Needle Retention 66
Ambient Air Pollution Concentrations in Indiana 66

DISCUSSION 69
CONCLUSIONS 71
REFERENCES 74

SECTION V - CALCULATION OF YIELD-LOSS COEFFICIENTS FOR MAJOR CROPS IN THE OHIO RIVER BASIN

INTRODUCTION 77
REVIEW OF LITERATURE 77
Yield Loss and Visible Leaf Injury 78
Environmental Conditions and Pollution Sensitivity 79
Interacting Effects of More Than One Pollutant 80
The Lower Limit for SO_2 Exposures Causing Crop Yield Reductions 81
Cumulative Exposure to O_3 and SO_2 as a Predictor of Crop Losses 82

CALCULATION OF CROP LOSS COEFFICIENTS FROM O_3 IN THE OHIO RIVER BASIN 84
Estimating the Effects of Cumulative O_3 Exposure on Soybean Yield 84
Calculation of O_3 Loss Coefficient for Soybean Yield 89
Calculation of O_3 Loss Coefficient for Corn 92
Calculation of O_3 Loss Coefficient for Yield of Wheat 96

O_3 LOSS FUNCTIONS RELATIVE TO AMBIENT AIR QUALITY STANDARDS 97
CALCULATION OF ESTIMATED CROP LOSS COEFFICIENTS FROM SO_2 IN THE OHIO BASIN 98
Effects of SO_2 Exposure Upon Soybean Yield 98
Calculation of SO_2 Soybean Yield Loss Coefficients 103
Effects of SO_2 Exposure Upon Wheat Yield 106
Calculation of SO_2 Yield Loss Coefficients for Wheat 109
Calculation of SO_2 Yield Loss Coefficients for Corn 111

CONCLUSIONS 112
REFERENCES 115

SECTION VI - CALCULATION OF AREAS OF DIRECT SULFUR DIOXIDE IMPACT AROUND COAL-FIRED POWER PLANTS

INTRODUCTION 121
SO_2 CONCENTRATION ISOPLETHS AROUND POWER PLANTS WITH MONITORING NETWORKS 122
SO_2 CONCENTRATION ISOPLETHS AROUND POWER PLANTS WITHOUT MONITORING DATA 144
DISCUSSION OF SO_2 DISTRIBUTION MODEL 162
CONCLUSIONS 164
REFERENCES 165
APPENDIX A 166

SECTION VII - CALCULATION OF TOTAL PRODUCTION LOSS OF SELECTED CROPS DUE TO DIRECT IMPACT OF SULFUR DIOXIDE

INTRODUCTION 173
METHODS 174
RESULTS AND DISCUSSION 176
CONCLUSIONS 181
REFERENCES 183

SECTION VIII - CROP LOSSES DUE TO OXIDANTS CAUSED BY EMISSIONS FROM COAL-FIRED POWER PLANTS IN THE OHIO RIVER BASIN

INTRODUCTION 185
PRESENT CONCENTRATIONS OF O_3 IN THE OHIO RIVER BASIN 186
THE SOURCES OF O_3 191
    Natural Background 191
    The Anthropogenic Sources of O_3 191

NITROGEN OXIDES AND HYDROCARBONS AS RATE LIMITING FACTORS IN RURAL AREAS 194
    The Sources and Quantities of Hydrocarbons 194
    The Sources of Elevated NO_x Over Rural Areas 196
    Hydrocarbon and NO_x Ratios 198

THE POTENTIAL FOR UTILITY NO_x EMISSIONS TO CONTRIBUTE TO O_3 199
THE POSSIBILITY OF FUTURE INCREASES IN O_3 201
CONCLUSIONS 210
REFERENCES 214

SECTION IX - ESTIMATING FOREST GROWTH LOSSES DUE TO OXIDANTS AND POLLUTANT-INSECT INTERACTIONS

INTRODUCTION 217
BACKGROUND ON FOREST GROWTH EFFECTS 217
ESTIMATING TOTAL FOREST GROWTH REDUCTION IN THE OHIO RIVER BASIN 223
    1976 Baseline Estimates and Rationale 223

Estimation of Growth Losses for Softwoods 225
Estimation of Air Pollution Losses for Hardwoods in the Ohio River Basin 225
Summary of Softwood and Hardwood Growth Reductions for 1976 226

ESTIMATED FOREST LOSSES FOR THE YEAR 1985 227
ESTIMATED FOREST LOSSES FOR THE YEAR 2000 227
ESTIMATING THE EFFECTS OF INSECT/AIR POLLUTION INTERACTIONS 229
LOSSES OF HORTICULTURAL PLANTINGS 231
CONCLUSIONS 232
REFERENCES 235

SECTION X - EPILOGUE: ESTIMATING THE AGGREGATE EFFECT OF AIR POLLUTANTS AND ACID DEPOSITION

AGGREGATE EFFECTS OF THE AIR POLLUTANTS 237
AN AGGREGATE ACID DEPOSITION RESPONSE 243
Limited Objectives for an Interim Acid Rain Assessment 243
The Phenomenon of Acidic Deposition 245
Present and Projected Combustion Emissions 246
Data Based on Recent H+ Inputs 246
Inventory of Present Effects on Natural Resources 251
Trends for Future Lake and Stream Responses 256

CONCLUSIONS 263
REFERENCES 266

TABLES

III-1	Summary of papers reporting O_3 induced effects on selected plants at low to moderate concentrations.	22
III-2	Metabolic and biochemical factors increased by ozone concentrations at or below 0.15 ppm for various exposure periods.	25
III-3	Metabolic and biochemical factors decreased by ozone concentrations at or below 0.15 ppm for various exposure periods.	26
III-4	Sulfur dioxide effects--alone and in combination with other pollutants--to selected plants at or below 0.30 ppm.	30
IV-1	Location of white pines sampled.	49
IV-2	Disease index for air pollution symptoms on eastern white pine.	51
IV-3	Index of foliar injury for 1979 and 1980 calculated from mottling/flecking, necrosis, and needle retention categories rated on a 20 to 100 point scale. Index range is 80 to 400, with 80 = no visible disease and 400 = severe disease.	56
IV-4	Percent of eastern white pine at 9 stands in Indiana and 4 in Wisconsin with various symptoms attributable to air pollution.	57
IV-5	Multiple range analysis of site means in 1980 using Duncan's (DUN) and Student-Newman-Keuls (SNK) methods.	58
IV-6	Number of days with ozone episodes at or greater than 0.08 ppm at five sites in Indiana and two sites in Wisconsin, and number of hours over 0.08 and 0.10 ppm during May-August 1978-1979.	59
IV-7	Ozone and sulfur dioxide concentrations at key midwestern U.S. locations.	68
V-1	Effects of chronic ozone exposure upon soybean yield. Data are from chronic exposure studies using concentrations which have been measured in the Ohio River Basin.	88
V-2	Cumulative exposure to SO_2 and O_3 at monitoring stations in the Ohio River Basin.	91

V-3	Effects of chronic ozone exposure upon yield or growth of *Zea maize* (corn) *Triticum sativum* (wheat), and *Lolium multiflorum* (ryegrass).	94
V-4	Effects of chronic SO_2 exposure upon soybean yield. Data are from chronic exposure studies using concentrations which are reached near power generating stations in the Ohio River Basin.	101
V-5	Effects of chronic exposure to SO_2 upon dry weight of *Lolium perenne* L., and *Triticum sativum*. Data are from chronic exposure studies using concentrations which are reached near power generating stations in the Ohio River Basin.	108
V-6	Estimated crop loss coefficients for chronic SO_2 and O_3 exposure. The basis for each of the loss coefficients is given in the text.	113
VI-1	AEP power plants with SO_2 monitoring networks.	123
VI-2	Emissions, Gaussian and Poisson distribution constants, and major and minor radii for five studied Ohio River Basin power plants.	138
VI-3	Stack heights and isopleth areas of five studied Ohio River Basin power plants.	148
VI-4	Sulfur oxide emissions and calculated concentrations isopleths of power plants in the Orbes Region for 1976.	151
VI-5	Summary of the area within the calculated concentration isopleths expressed as a percent of the total area of the state within the Orbes Region.	161
VII-1	Example calculation of estimated production loss caused by SO_2 in three crops around a single coal-fired power plant.	175
VII-2	Estimated increases of soybean production due to abatement of peak-load SO_2 emissions from coal-fired electrical generating stations in the ORBES Region.	177
VII-3	Estimated increases of corn production due to abatement of peak-load SO_2 emissions from coal-fired electrical stations in the ORBES Region.	178
VII-4	Estimated increases of wheat production due to abatement of peak-load SO_2 emissions from coal-fired electrical generating stations in the ORBES Region.	179

VII-5 Estimated production increases of three crops due to abatement of peak-load SO_2 emissions from coal-fired electrical generating stations. 180

VIII-1 NO_x concentrations at four rural sites in the vicinity of coal-fired power plants in the Ohio River Region. 197

VIII-2 Sources of oxides of nitrogen in four states in the Ohio River Basin from 1973-1975. 200

VIII-3 Nitrogen dioxide trends in annual arithmetic mean, 1970-76. 203

VIII-4 Estimated NO_x emissions, O_3 changes, and percent of O_3 attributable to power plant emissions in the ORBES Region. 204

VIII-5 Crop loss functions for ozone effects on the species in the Ohio River Basin (%). 208

VIII-6 Total gain in production (10^6 bushels) of three crops if oxidants originating from all sources in the ORBES Region were abated. 209

VIII-7 Current and projected reductions in annual yield from ozone/oxidants attributable to Ohio River Basin power plants (10^6 bu./yr.). 211

IX-1 Total forest growth reduction in the Ohio River Basin due to air pollutants, expressed in millions of cubic feet of wood. Total normal yield for the region in 1976 was 1.49×10^9 cubic feet. 228

IX-2 Total horticultural losses in the ORBES Region due to pollutant and insect/disease interactions. 233

X-1 Estimated crop yield benefits expected from abatement of sulfur dioxide emissions from coal-fired electrical generating stations, 1976, and no-compliance scenario, 1985 and 2000. Gain is for entire ORBES Region. 238

X-2 Estimated range of total losses for three crop species due to ozone and associated oxidants formed from ORBES power plant emissions, 1977, 1985 and 2000. 241

X-3 A consensus on present scientific understanding of atmospheric acid formation, sources, and deposition. 248

X-4 Consensus on present scientific understanding of atmospheric acid effects on plants. 253

X-5 Consensus on present scientific understanding of atmospheric acid effects on aquatic organisms. 257

FIGURES

III-1 Comparison of federal ambient air quality standards for SO_2 doses that produced significant plant and ecosystem stress. 29

IV-1 Location of nine eastern white pine stands surveyed in 1979 and 1980. 48

IV-2 Location of eastern white pine stands around the Columbia Generating Station in Wisconsin. 50

IV-3 Frequency histogram of disease index scores for Indiana sites during 1979 and 1980 and four Wisconsin sites in 1980. 61

IV-4 Linear regression of the coefficients of skewness (g_1) for the symptom of flecking/chlorotic mottling and disease indices for 13 sites in Indiana (1979-1980) and Wisconsin (1980). 62

IV-5 Linear regression of skewness coefficients (g_1) for percent of needle necrosis and disease indices for 9 sites in Indiana (19791980) and 4 sites in Wisconsin (1980). 64

IV-6 Linear regression of skewness coefficients (g_1) for length of needle necrosis and disease indices. 65

IV-7 Linear regression of skewness coefficient (g_1) for needle retention and disease indices at sites in Wisconsin and Indiana. 67

IV-8 Areas around coal-fired power plants where calculated SO_2 concentrations are sufficient, in conjunction with regional ozone concentrations, to cause slight-to-moderate disease injury to tolerant white pines, and severe disease injurty to sensitive white pines. 72

V-1 Maximum hourly ozone averages for June, July, and August 1977 at three urban stations in the Ohio River Basin. 85

V-2 Profile of daily ambient ozone at Indianapolis and Raleigh North Carolina. 87

V-3 Effects of chronic ozone exposure upon soybean yield (seed or pod weight). 90

V-4 Effects of chronic ozone exposure upon yield of corn. 95

V-5 Cumulative frequency curves for SO_2 monitoring stations inside Impact Zone I at Clifty Creek generating station. 100

V-6 Effect of chronic SO_2 exposure on soybean yield. 104

VI-1 AEP monitoring network around the Gavin-Kyger Creek-Sporn power plants. 125

VI-2 AEP monitoring network around the Clifty Creek power plant. 126

VI-3 AEP monitoring network around the Breed power plant 127

VI-4 AEP monitoring network around the Muskingum River power plant. 128

VI-5 AEP monitoring network around the Big Sandy power plant. 129

VI-6 Means of the second-highest hourly SO_2 concentrations ($\mu g/m^3$) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Gavin-Kyger Creek-Sporn power plants. 130

VI-7 Means of the second-highest hourly SO_2 concentrations ($\mu g/m^3$) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Clifty Creek power plant. 131

VI-8 Means of the second-highest hourly SO_2 concentrations ($\mu g/m^3$) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Breed power plant. 132

VI-9 Means of the second-highest hourly SO_2 concentrations ($\mu g/m^3$) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Muskingum River power plant. 133

VI-10 Means of the second-highest hourly SO_2 concentrations ($\mu g/m^3$ during the months of June, July, August (1975-1977) for monitoring stations (o) around the Big Sandy power plant. 134

VI-11 Comparison of two methods of summarizing monitored SO_2 concentrations around power plants. 135

VI-12 Comparison of two methods of summarizing monitored SO_2 concentrations around power plants. 136

VI-13 Gaussian curves used to describe pollutant concentrations in the crosswind (Y-Y') direction. 139

VI-14 Poisson curves used to describe pollutant concentrations in the downwind (X-X') direction. 140

VI-15 Calculated SO_2 concentration isopleths for the Gavin (G)-Kyger Creek (K)-Sporn power plants. 143

VI-16 Relationship between emission rate and major radii (Poisson distribution) of the 0.1-ppm and 0.05-ppm isopleths. 145

VI-17 Relationship between emission rate and minor radii (Gaussian distribution) of the 0.1-ppm and 0.05-ppm isopleths. 146

VI-18 Projections of nominal NO_x and SO_x emissions (1980-2000 under three scenarios of utility development in the Ohio River Basin. 150

VI-19 1976. Calculated impact areas of all power plants in the ORBES region with SO_2 emission rates of greater than 2000 g/sec. 153

VI-20 1985. Scenario 2, assuming compliance with SIP's by 1985. 154

VI-21 1985. Scenario 2d, assuming no compliance with SIP's. 155

VI-22 1985. Scenario 7, assuming compliance with SIP's. 156

VI-23 2000. Scenario 2, assuming compliance with SIP's by 1985. 158

VI-24 2000. Scenario 2d, assuming no compliance with SIP's. 159

VI-25 2000. Scenario 7, assuming compliance with SIP's. 160

VIII-1 Surface analysis of the movement of air masses across the easterm United States containing O_3 exceeding 70 ppb from 4 September through 7 September 1976. 187

VIII-2 Ozone concentration profile in the area of Indianapolis, Indiana, at three dates in 1975-76. 189

VIII-3 Ozone concentration profile at Indianapolis, Indiana, at two dates in 1975-76. 190

VIII-4a Aerometric data showing the concentration of SO_2 particles larger than 0.01 μm, and O_3 in a power plant plume traveling 300 m above Lake Michigan. 192

VIII-4b Trends in the average O_3 concentrations as observed in the air downwind of a power plant. 193

VIII-4c Theoretical predictions of the excess O_3, HNO_3, and sulfate aerosol in the air downwind of a power plant. 193

VIII-5 Map based on LANDSAT-1 data showing areal extent of smoke plumes originating at Philo and Beverly, Ohio. 195

VIII-6 Trends in O_3 standard violations at selected sites in California and the nation, 1973-1977. 202

VIII-7 Trend in episode ozone concentrations in Ohio River Basin from 1929 to 2000 A.D. 205

IX-1 Calculated average cross-sections of two 30-year-old ponderosa pines at breast height grown in polluted air (left) and in non-polluted air (right) based on radial growth samples from 1941-1971 and 1910-1940. 218

IX-2 Relationship of precipitation, oxidant dose and foliar injury to radial and vertical growth of ponderosa pine saplings before and after treatments in filtered (FAH) or unfiltered (AAH) air greenhouse and an outside ambient air treatment (AAO). 219

IX-3 Relation of reduced ring widths of eastern white and loblolly pines to increased air pollution from a Virginia munitions plant over a 3-war period of 1935 to 1975. 221

IX-4 Ring-width of trees downwind from a munitions plant in central Wisconsin in four grouped time intervals from 1964 to 1977. 222

X-1 Reduction in soybean yield (%) projected for the Ohio River Basin under conditions of high electrical growth. 242

X-2 Historic trends in agricultural crop yields per hectare in Michigan between 1880 and 1978. 244

X-3 Flow diagram showing system linkages for acid precipitation formation, deposition, and effects as a consequence of nitrogen and sulfur oxide emissions from fossil-fuel combustion. 247

X-4 Projections of NO_x and SO_x emissions (1980-2000) under three scenarios of utility development in the Ohio River Basin. 249

X-5 Precipitation pH isopleths from 1955-1956 to 1972-1973 showing the time-course of change in H+ loading for the northeastern United States (Cogbill and Likens 1974). 250

X-6 Historical background (ca. 1850), recent observations (1965-1977), and projections (1980-2000) of total annual deposition of sulfate plus nitrate. 252

X-7 Frequency distribution of pH in lakes in the Adirondack Mountains, New York, in the 1930's. 255

X-8 Measurements of pH in waters of the Tusket River of Western Nova Scotia between 1955 and 1975. 258

X-9 Relationship between acid loading and pH change for lakes in Sweden in very sensitive (1) and somewhat less sensitive (2) surroundings. 259

X-10 Estimate of total number of lakes in the Province of Ontario susceptible to acidification by the year 2000, based on present alkalinities, levels of currently-known acid inputs, and a generalized acidification relationship. 261

X-11 Probable response times for lakes of Ontario, Quebec, New York, New England, and the Maritime Provinces subjected to acidic deposition. 262

SECTION I

INTRODUCTION

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Adjusting to national energy and environmental needs has led industry and state and national governments to evaluate fully the feasibility and environmental consequences of greatly increased coal utilization. In the coal-rich Ohio River Valley area, public utilities announced plans for a large number of coal-fired power plants following the 1973-74 Arab oil embargo. Because of increasing prices and uncertain supplies of oil and gas, and because coal and water are abundant in the Ohio Valley, a major shift to coal seemed likely in this area by the year 2000. These were the basic reasons for undertaking the intensive Ohio River Basin Energy Study (ORBES) of which this report on ecological effects is a part.

Decisions about future power plant sites will, however, have to balance many interests in addition to energy supplies, water, and costs. In Minnesota, farm groups believe that agricultural yields have been diminished noticeably in areas around major power plants. State legislation has been introduced to limit agricultural losses by prohibiting development of power facilities in excess of 1000 megawatts in any one county.[1] The possibility of adverse consequences for crops and forests in the Ohio River Basin seemed to warrant a thorough evaluation as part of any analysis of power plant fuels and sites in the region.

Because of the concerns about effects from greatly expanded coal utilization in the Ohio Valley area, Congress funded, and the U. S. Environmental Protection Agency has implemented the Ohio River Basin Energy Study through the cooperation of nine Midwestern universities.[2] The ORBES study was designed to help industrial and public planners determine a "consequences-versus-benefits" balance for the projected regional expansion of generating capacity and of the coal-extraction industry. The support study undertaken by TIE for ORBES was entitled "Subinjurious Effects of Gaseous Sulfur and Nitrogen Emissions and Their Conversion Products on Crops and Forests in the Ohio River Basin States."

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[1] Bill S.F. No. 793, Minnesota Legislature, invalidating certificates of need for certain large electric generating plants.

[2] For details of other studies, see the ORBES Summary report "Ohio River Basin Energy Study (ORBES) Main Report" and individual reports on each project as cited in the main report. Office of Research and Development, U.S. Environmental Protection Agency.

Sulfur dioxide (SO_2), oxides of nitrogen (NO_x), and ozone (O_3), alone or in combination at only moderate concentrations can produce effects on many plant species. At concentrations now being observed with some frequency in the ORBES area, these gases can reduce the growth, yield, or quality of plants without producing the recognized injury symptoms. Thus, although "physiological injury" can be documented, the expression "subinjurious effects" has come into frequent usage for these subtle plant responses.

In the ORBES region a relatively large part of the sulfur and nitrogen oxides are produced by the burning of coal (see last section). Many of the previous attempts to assess effects of these combustion emissions have been limited to experimental (laboratory) or site-specific (one power plant area) studies, and to "observable" injury on the plants. However, recent studies reviewed in this report have shown that with long-duration chronic exposures, substantial crops losses may occur before the usual injury symptoms are apparent. Credible evidence, based largely on occasional surveys of observable injury, suggests that crop and forest losses from all fossil fuel emissions might be significantly higher than conventional estimates. This study by TIE was undertaken to consider all available evidence of possible crop losses, both at present, and under future coal development scenarios.

Thus, the specific objectives of this study of gaseous sulfur and nitrogen emissions effects, and of the effects of their transformation products (ozone and acid rain) on crops and forests of the Ohio River Basin are:

(1) To summarize and evaluate the literature on metabolic responses of plants and their insect pollinators to sub-injurious levels of sulfur oxides (SO_x), their acidic conversion products, acid particulates, and oxidants;

(2) To assemble data on concentrations of emissions around representative primary emission sources in the region;

(3) To determine the distribution and compute the acreage of sensitive crop and forest species in the areas near emission sources;

(4) To review and apply damage functions for the sensitive species to computation procedures for estimating losses in yield from a range of current and projected pollutant levels;

(5) To estimate, in a format that can be applied to an economic evaluation of losses, the probable maximum and probable minimum

effects of the emissions and transformation products on the regional production of crop and forest products; and

(6) To incorporate a review, by outside scientists and policy analysts, of the data base and computation procedures employed and the conclusions reached during the study.

Estimating crop production losses due to chronic exposures in the field is almost impossible with experimental data obtained in controlled situations on a one-pollutant basis. Some recent field experiments, however, have considered all three phytotoxic gases of interest (SO_2, NO_x, and O_3). The timing of the brief exposures, synergisms among the toxicants, weather conditions, and crop varieties greatly influence effects. Direct measurement of yield effects are possible only through carefully designed field or mixed-pollutant experiments, but these are now becoming available. In this study, therefore, data from field and laboratory experiments will be considered in conjunction with effects modeling procedures to estimate overall regional consequences for forest and crop production.

The experimental design for the study is structured around several scenarios for regional energy development between 1976 and 2000 as developed by the ORBES research team. Two scenarios assume a near-doubling of the region's electrical generating capacity between 1976 and 2000, one with full compliance to Clean Air Act regulations, and one without compliance (scenarios 2 and 2d, respectively). It is assumed that existing coal-fired power plants will be retired and replaced gradually by smaller, cleaner coal-fired plants. A third scenario (Scenario 7, High Electricity Growth) assumes a longer power-plant life -- 45 years -- rather than 35 years as assumed in the other scenarios. For the most part, the increased growth in electricity use by the year 2000 is to be met by construction of numerous moderate-sized, low-sulfur coal-fired power plants after 1989.

The goals and structure of ORBES, itself an analysis of development options, using the best information available, have necessarily had a major influence on how the TIE researchers could best proceed to meet the objectives of their support study. When technical information is used in examining policy options, the research questions are not so much about scientific certainty (although this is important) but about what the experimental data-base already available says (on crop and forest production) for the region of interest (the Ohio Valley states). The ORBES study, in turn, sought to use output from the TIE study to evaluate short- and long-term economic implications of the development options represented in the various scenarios. Thus, the effects information had to be cast in terms of state and regional agricultural (and forest) production, insofar as possible.

Although, in a narrow sense, much of the effects data-base available to TIE was designed to test explicit study hypotheses different from those examined here, in a broader sense these studies were funded to make possible evaluation of the types of development options posed in the ORBES scenarios. Thus, the original research undertaken here concerns the methodologies of information synthesis and transfer to new contexts. Previous work of this type often has been carried out as part of economic evaluations ("economic damage functions"), but in the work reported here, each aspect of the pollutant effects process has been evaluated before the econometric techniques are applied.

To meet these goals, the report has taken the following structure: following introductory and summary sections, a brief literature review on the mechanisms of phytotoxic gaseous effects; a report on field observations of pollution-related foliage symtoms in a cross-section of the ORBES region; an intensive review and synthesis of published data in the form of regional "crop damage functions" for SO_2 and O_3; computation of areas of elevated SO_2 exposures around present and proposed generating stations; computation of projected yield reductions for the acreages exposed to moderate O_3 concentrations, estimation of forest and horticultural effects; and finally, an epilogue which considers all of the effects from within the ORBES region with what we know about effects from acid rain outside the region that may be attributable to ORBES area policies.

An issue of much importance to the authors throughout this study has been the question of legitimate disagreement among scientists and in the recent published literature. Some researchers hold strongly to the view that as long as doubt remains in the view of even a few qualified scientists, no consensus statements are warranted, other than recommendations for additional research. However, this study was conducted in the context of technical and policy analyses relating to immediate decisions on regional energy development. Thus, the authors have had an obligation to incorporate an applied approach in their work. We have, therefore, used the scientific evidentiary standards of administrative and judicial proceedings, i.e. simply a preponderance of credible scientific opinion. Thus, the findings of fact and findings of probable or possible relationship reported here reflect what is judged by the authors to represent a preponderance of present evidence, likely to stand up under cross-examination against countervailing studies or opinion.

Inevitably, there exists some doubt, and the findings may be subjected to modification upon presentation of new research results when the technical and scientific issues become more fully explored. Obviously, on some questions left unanswered here, scientific differences will continue and uncertainty will prevail. For the present, however, the investment in pollutant effects research over the past 30 years appears to have been sufficient for examining the principal issues of the Ohio River Basin Energy Study.

SECTION II

SUMMARY AND FINDINGS

The U. S. Environmental Protection Agency in 1976 was directed by Congress to carry out a comprehensive assessment of the regional consequences of major proposed expansion of coal-fired electrical generating capacity in the Ohio River Basin. The "Ohio River Basin Energy Study" (ORBES) was being carried out by a consortium of nine Midwestern universities, with support studies at other institutions, including the work reported here by The Institute of Ecology (TIE). Several parts of the ORBES program itself, and support studies such as the atmospheric transport modeling undertaken by Teknekron, Inc.,[1] have defined the regional development options within which to undertake detailed evaluation of resource productivity effects from greatly increased coal combustion in the Ohio River Basin states.

The portion of the ORBES project reported here, entitled "Subinjurious Effects of Gaseous Sulfur and Nitrogen Emissions and Their Conversion Products on Crops and Forests in the Ohio River Basin States," was conducted in three phases between February 15, 1979, and June 15, 1980. The TIE support study included several major objectives sought by the study as a whole, and the principal sections of the report reflect those divisions.

The most important objectives of the study can be summarized as:

- Summarization and evaluation of the literature on metabolic responses of plants to low levels of sulfur oxides (SO_x), their acidic conversion products and oxidants, particularly ozone (O_3)
- Assembling data on concentrations of emissions around representative primary emission sources in the region;
- Determine the distribution and acreages of sensitive crop and forest species in the areas near emission sources;
- To review and apply damage functions for key sensitive species to estimate losses in yield from a range of current and projected pollutant levels;

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[1] For details of other studies, see the ORBES Summary report "Ohio River Basin Energy Study (ORBES) Main Report" and individual reports on each project as cited in the main report. Office of Research and Development U.S. Environmental Protection Agency.

- To estimate, in a format that can be applied to an economic evaluation of losses, the probable maximum and probable minimum effects of the emissions and transformation products on the regional yield of crop and forest products.

The problems inherent in taking research results obtained in site-specific experimental situations and adapting them to a regional policy question are reviewed in the Introduction. Obviously, the data were originated for just such situations, but there is not yet a broad concensus on the transfer and application process.

A review of literature on the mechanisms of action of gaseous pollutants on crops and forests, (Section III) summarizes recent work relating to the magnitude of effects for trees and crops due to SO_2, NO_x, and O_3. Section III findings include:

Factual

- Experiments on many plant species show that SO_2 exposures of from 1 to 4 hours at 0.10 to 0.30 ppm can cause injury or growth related effects. These concentrations are well below the current federal SO_2 standard of 0.5 ppm for three hours. Synergistic reactions between SO_2, O_3, and NO_2 and other gases, although not universal are now being frequently documented.

- Sulfite has been identified as the chemical form largely responsible for causing SO_2-related injury within the plant system. Sulfite reacts with cellular components including proteins, enzymes, aldehydes, ketones and olefins. Studies have shown that SO_2 exposures for several hours at up to 0.15 ppm result in physiological disturbance such as plasmolysis, inhibition of photosynthesis, depression of growth rate, increased respiration and enhanced leaf senescence.

- Ozone at concentrations of 0.05 to 0.08 has been shown to cause effects including decreased photosynthetic rate and carbohydrate formation, membrane disruptions, enzyme coagulation, osmotic dislocations, lowered ATP production, premature leaf senescence, disruption of citric acid cycle, disruption of organelles, and inhibition of the partitioning of solutes to roots, shoots and fruits.

- Yield losses of crops occur at O_3 concentrations that do not cause visible injury in the species affected.

Section IV, entitled "Assessment of Air Pollution Injury to Eastern White Pine in Indiana" is a field study of foliage condition on eastern white pine, a sensitive native conifer of importance to forestry and horticulture. The study was carried out in central and southern Indiana, and in Wisconsin as a possible control area, to evaluate responses of a sensitive native species in the region. Findings include:

Factual

- Prior research has shown eastern white pine is sensitive to O_3 and SO_2, both separately and together. By careful observation on the effects of chronic or acute exposures to these pollutants can be distinguished from the effects of other damaging agents.

- Ozone measurements in the ORBES region indicate that current concentrations are sufficient to cause chronic damage to this sensitive species. Potentially damaging concentrations of SO_2 occur more locally around utility and industrial sources in Indiana.

- Symptoms of air pollution disease were found at all sites sampled in Indiana and Wisconsin and more than 90% of all trees showed symptoms. More than 80% of the trees sampled in Indiana had needle necrosis on 20% or more of the foliage; only one-fourth of the trees in Wisconsin were similarly affected.

- A disease index based on all measures of air pollution damage shows that air pollution disease is only slight in Wisconsin and moderate to severe in Indiana, depending on local pollutant concentrations.

Probable

- The chronic damage found on trees in Wisconsin and Indiana is probably due to the high regional O_3 concentrations. The greater severity of disease in Indiana appears to result from locally combined SO_2 and ozone exposure.

- Chronic exposure to presently occurring O_3 concentrations appears to be weakening the trees, making them more susceptible to other environmental stresses.

Section V, entitled "Calculation of Yield-Loss Coefficients for Major Crops In The Ohio River Basin" uses the published experimental work in yields of soybeans, corn and wheat under controlled SO_2 and O_3 exposures to obtain crop loss coefficients for these species. The data come from studies conducted in the field and laboratory, with the known shortcomings of both approaches taken into account. The crop loss estimates range from lower bound, or minimum values, representing the most conservative assumptions possible, to upper bound, or maximum values, reflecting likely aggregate plant response during sensitive growing conditions and life-cycle stages. Findings include:

Factual

- Recent evidence, from chronic exposure studies, shows yield or growth reduction in many species occurs without visible injury. There is little experimental basis for the assumption that crop damage due to chronic exposures can be predicted from the leaf injury or short term reduction in photosynthesis caused by acute exposures.

- Ambient O_3 and SO_2 concentrations being observed at monitoring sites in the Ohio River Basin are at levels which, under experimental conditions, have caused reductions in crop yields.

- Studies have shown that cumulative exposure, expressed as the sum of concentration above some background level times the duration of exposure (in hours), rather than acute exposures, is the best expression for estimating yield loss. The same approach is used to assess both O_3 and SO_2 damage, although high peaks characteristic of SO_2 fumigations can cause mild injury that is not estimated by summing hourly averages of SO_2 exposure.

Probable

- Periodic occurrences of SO_2 concentrations at or above 0.05 ppm are considered to be sufficient to reduce growth and yield of sensitive species, when in the presence of other phytotoxic gases such as O_3 and nitrogen oxides.

- Evidence strongly suggests that the presence of two or more pollutants in a crop environment can cause additive or synergistic effects. Some of these pollutant interactions are assumed to have been reflected in the field experimental studies where only one gaseous pollutant is manipulated in a background of ambient pollutants. Several studies support a conclusion that effects observed on some species under SO_2 treatments as low as 0.05 ppm are partly due to the presence of background concentrations of O_3 between 0.05 and 0.10 ppm.

- The potential fertilizing effect of atmospheric sources of sulfur has been discounted for the Ohio River Basin because agricultural soil sulfur levels in this area are not limiting and nitrogen fertilization apparently is not inducing sulfur deficiency.

- Yield loss functions for O_3 based on numerous controlled field studies throughout the humid eastern U.S. indicate a lower bound of 5% and an upper bound of 26% annual loss for soybeans, depending on seasonal concentrations of O_3 and other weather-related factors, and utilizing cumulative O_3 exposure (concentration hours) from O_3 monitoring at various locations within the ORBES region. Although the loss function itself is based on an accumulation of data which form a strong trend-line (with transverse variance) the dose ($\mu g \cdot m^{-3}h$) is based on monitoring in the ORBES ares.

- Loss functions for O_3 effects on corn range from a lower bound of 4% to an upper bound of 15% depending on exposure and variety, for the same field exposures monitored in the ORBES region.
- Loss functions for O_3 effects on wheat range from a lower bound of 3% to an upper bound of 10%.
- Yield loss coefficients for soybeans due to the SO_2 exposures observed in the vicinity of major power plants range from a lower bound of 1% to an upper bound of 9% as a loss over and above O_3 related effects on the same plants. These are based largely on field experiments where the SO_2 concentrations were similar to those monitored around power plants.
- Although data are minimal, and the impacted areas are modest, short-term studies suggest that chronic SO_2 effects on corn cannot be discounted. Low concentrations of SO_2 can cause reduced stomatal resistance and increased water loss. The data available indicate a loss function ranging from 0% to 3% for the exposures being observed in the vicinity of power plants in the ORBES area.
- The SO_2 loss functions for wheat are based on studies of Lolium, a related genus, The data indicate a range from a lower bound of 1% to an upper bound of 6%.

Research results which support the findings of ozone-induced yield losses presented here come from studies of corn, soybeans, potatoes and tomatoes in eastern states outside the ORBES region. Yield reductions for these species, similar to or greater than losses estimated for the Ohio River Basin, have been reported to be caused by ambient ozone exposures with high humidities similar to those in the region. Further statistical analyses may refine these loss coefficients but complete testing of the assumptions underlying the analyses would require an improved data base within the ORBES states. Studies should be conducted in the field with realistic pollutant exposures and statistically sound experimental designs.

The area affected by SO_2 emissions from power plants was estimated (Section VI) using a field data-based model and complete emissions inventories for the years 1976, 1985 and 2000 supplied by Teknekron, Inc., as their part of the ORBES work. The area around each coal-fired generating station where SO_2 concentrations could cause crop losses (greater than

0.05 ppm SO_2 at least twice per month) was estimated using a simple ellipse model based on observed relationships between the SO_2 monitoring data and utility SO_2 emission rates.

Future SO_2 emission projections were provided to the TIE researchers in accordance with several energy development scenarios adopted by the ORBES core term. Scenario 2 assumes that utilities will comply with State Implementation Plans (SIPs) for pollutant control and that electrical generating capacity will double by the year 2000. In Scenario 2d the rate of electrical growth is the same but no compliance with SIPs is assumed. Both of these scenarios assume a 35-year power plant life and new generating capacity consisting of smaller, cleaner coal-fired plants (650 megawatts each) and a few nuclear plants. A third projection (Scenario 7) assumes extended power plant life (45 years) and complicance with SIPs. Total generating capacity under Scenario 7 in 2000 is about 15% higher than under the other two scenarios. Sulfur oxide emissions in 2000 are about 40% higher in Scenario 2d (lax compliance) compared to Scenario 2; Scenario 7 is intermediate. Findings include:

Factual

- Average second-highest monthly SO_2 concentrations were observed to decrease gradually downwind, i.e. northeast of the power plants. The data are normally distributed in the crosswind direction and follow a Poisson distribution downwind. Power plants with an emission rate below 2000 g/sec did not appear to produce an impact area with concentrations greater than 0.05 ppm SO_2.

- The area of elevated SO_2 concentrations is affected primarily by the magnitude of emission rates, rising in an exponential fashion. No explicit relationship was found between stack height and elevated SO_2 distribution around the power plant, although this is probably due to the intercorrelation between emission rate and stack height.

Probable

- The observed SO_2 distribution is probably due to pollutant trapping in the lower layers of the atmosphere following the slow passage of stagnant high-pressure systems.

- In 1976, an estimated 9.3% of the entire ORBES region was affected by peak SO_2 concentrations (expressed as the average of the June, July, and August second-highest hourly observation) at more than 0.05 ppm SO_2.

- Under the emissions and control assumptions of the three ORBES scenarios examined in this study, non-compliance (Scenario 2d) leads to a total area of 9.6% impacted by SO_2 in 1985, and 6.0%

in 2000. Compliance (Scenario 2) is projected to lower the affected area to 3.1% in 2000.

- Under the assumption of high energy growth (Scenario 7) the area expected to be impacted by elevated SO_2 concentrations will be 4.7% of the ORBES region in 1985 and 4.5% in 2000.

- Various considerations support a conclusion that the calculated concentration isopleths are not in error by more than 10% in homogeneous topography, and that there is no significant bias.

Sections VII, VIII and IX of the report contain calculations of the aggregate (season-long) effects on growth (forest and crop species)and yields (crop species only) within the ORBES region. These effects are expressed initially in terms of per-unit-area yield reductions and reduced longevity. As a second step, the apparent reductions in crop yields and forest growth, increases in early mortality of forest species, and effects horticultural materials were converted from unit-area estimates to total production effects by state.

In Section VII, region-wide SO_2-induced production losses for soybeans, corn and wheat were estimated by calculating the crop areas within the impact zone around individual power plants and applying the crop-loss coefficients to those acreages. The calculations retain the upper and lower bounds of the crop loss coefficients, and, owing to the nature of the best field SO_2 effects data, must be viewed as the incremental (additive or synergistic) effect of the SO_2 concentrations in a background of elevated O_3 and NO_x observed in the same area.

Annual yields from three years (1975-1977) for each crop were averaged to obtain a mean normal yield for each impacted county. Because this normal yield already includes the effects of air pollutants existing at the time, a probable "clean air yield" was calculated by increasing the normal yield in proportion to the SO_2 loss coefficients calculated in Section V. Crop loss estimates, therefore, represent the product of total crop area impacted by SO_2, the estimated clean air yield, and the crop loss coefficient. Total ORBES region production losses were obtained as the sum of individual site losses. Findings include:

Factual

- Total annual production in the ORBES region for corn (three-year average, 1975-1977) is 2.11×10^9 bushels, for soybeans it is 0.55×10^9 bushels, and for wheat 0.18×10^9 bushels.

Probable

- Crop production losses due to SO_2 exposures near power plants expressed as gains expected from SO_2 abatement, can be summarized as: in 1976, total gain for soybeans is projected to be in the range from 0.65 to 2.12 million bushels; for corn, from 0 to 3.53 million bushels; and for wheat, from 0.21 to 0.48 million bushels.

- Under the assumptions of Scenario 2 (SIP compliance), projected gains from complete abatement are (in millions of bushels) 0.26 to 0.83 in 1985, and 0.19 to 0.60 in 2000 for soybeans; for corn, 0 to 1.34 (1985) and 0 to 1.01 (2000); and for wheat, 0.07 to 0.17 (1985) and 0.06 to 0.14 (2000). Comparable projections for Scenario 2d (noncompliance) are 0.59 to 1.92 (1985) and 0.40 to 1.30 (2000) for soybeans; 0 to 3.24 (1985) and 0 to 2.42 (2000) for corn; 0.23 to 0.50 (1985) and 0.16 to 0.36 (2000) for wheat. Assuming high energy growth and extended power plant service life (Scenario 7), gains from SO_2 abatement are projected to be 0.25 to 0.81 (1985) and 0.22 to 0.73 (2000) for soybeans; 0 to 1.32 (1985) and 0 to 1.27 (2000) for corn; 0.08 to 0.17 (1985) and 0.07 to 0.16 (2000) for wheat, in millions of bushels.

- The aggregate reduction in agricultural production attributable to SO_2 in the areas of impact around power plants is estimated to be less than 1% of total regional production in 1976, and should decrease by the year 2000 in all of the development scenarios.

- From 40% to 60% (depending on energy development scenario) of the total potential ORBES gain in soybean production could be achieved by SO_2 emission controls at three large power plants in the important agricultural areas of Illinois. The same values for corn are 30% to 50%, and for wheat, 20% to 45%.

Section VIII examines regional O_3 induced crop losses, and considers the potential contribution of power plant emissions to the regional O_3 concentrations. A relatively persistent deep layer of O_3 is known to blanket much of the region during episodes favorable to O_3 formation and accumulation in the mid-west. Ozone losses to soybean, corn and wheat production (or gains from abatement) have been estimated by applying the O_3 crop loss coefficients (as developed in Section V) to all of the crop acreage in the region. The following are findings:

Factual

- Ozone concentrations monitored in rural regions of the Ohio River Basin are of sufficient concentration and duration to cause negative responses in vegetation and crops.

- Studies of coal-fired power plant plumes have shown that, under certain weather conditions, the emissions of nitrogen oxides reacting with local and regional sources of non-methane hydrocarbons produce 20% to 30% increases in ambient O_3 concentration.

- Observations of O_3 in the ORBES area indicate frequent elevated concentrations extending 7,000 to 9,000 feet into the lower atmosphere during the summer months. Overnight depletion of O_3 occurs only in the lower 500 feet of this layer.

- In the Ohio River Basin, O_3, NO_x, and hydrocarbons are present in rural areas, outside of well-defined industrial or urban plumes, at concentrations sufficient to enter into slow O_3-forming reactions.

- Total utility emissions of NO_x were 1.35 billion kilograms in 1976, about one-third of total NO_x emissions in the region.

Probable

- Because of the combination of naturally produced hydrocarbons from forests and agricultural crops in the Ohio River Valley and significant sources of anthropogenic hydrocarbons, this precursor appears to be rarely a rate-limiting factor for O_3 formation. Nitrogen oxides, which are produced in the Ohio River Basin region from urban sources as well as from power plant emissions, appear to be a more frequent rate-limiting factor.

- Pre-industrial ambient O_3 levels at low altitudes appear to have been low (0.03 to 0.05 ppm, except during episodes associated with thunderstorms or cyclonic activity).

- If O_3 formation in the ORBES area is frequently NO_x rate-limited (see above), then the current NO_x emissions from power plants could contribute 30% to 50% of the ozone in the region. This estimate assumes a 1:1 NO_x/O_3 production rate and the increased probability that NO_x emissions from elevated stacks will lead to ozone because ground-level NO_x is readily taken up by vegetation during the growing season period of interest to this study.

- The potential for future changes in O_3 concentrations are examined in relation to the historical and projected record of NO_x and hydrocarbon emissions. There is an apparent history of rising O_3 levels (from 0.03-0.05 ppm background) to current common episode levels of 0.10 to 0.15 ppm which is thought to account for the rapid rise in ozone pathologies of sensitive crops during the past 25 years. The apparent historical increase of about 0.02 ppm per decade appears likely to continue at least to 1985, if NO_x is the limiting component in O_3 formation in the mid-west, and the projections of 15% increases in NO_x emissions are accurate. After 1985, ozone levels are projected to increase more slowly, or decline, depending on NO_x emission controls and the various electricity generating scenarios.

- The ORBES development scenarios show utility NO_x emissions are likely to increase from 16% to 66% between 1985 and 2000, depending on the scenario, and assuming no NO_x abatement. These increases could be totally or partially offset by decreases in NO_x emissions from other sources; the assumption has been made that automobile emission controls and the trend to more fuel-efficient cars will offset in the order of 40% of the utility increase. This assumption suggests a decrease in O_3 levels with moderate electrical growth (Scenarios 2 and 2d), and more slowly rising O_3 levels with high electrical growth (Scenario 7).

- The decreased NO_x from non-utility sources leads to an increasing proportion of the O_3 being attributable to power plant sources by the year 2000. Projections of the proportion of O_3 due to power plant emissions in 1985 range from 30% to 50%. For the year 2000, O_3 attributable to power plants is projected to be 40% to 60% or 50% to 70% of total O_3, depending on the scenario.

- Crop loss functions in 1977, based on measured O_3 concentration and calculated concentration-hours of exposure, range from a lower bound of 5% to an upper bound of 26% for soybeans, 4% to 15% for corn and 3% to 10% for wheat (Section V). Crop loss functions in 1985 and 2000 are based on projected O_3 concentrations, and, therefore, projected cumulative concentration-hours of exposure. Soybean yield reductions are estimated to range from a lower bound of 24% to an upper bound of 41% in 1985 (all scenarios); corn yield reductions would range from 7% to no more than 18%; wheat yield reductions should range from 5% to 18%. An assumption of high electrical growth (Scenario 7) leads to projected yield reductions in 2000 of from 30% to 47% for soybeans; from 9% to 19% for corn, and from 7% to 23% for wheat. Under Scenario 2 and 2d, the assumption of reduced non-utility NO_x emissions leads to a considerable reduction in projected yield losses due to O_3 by the year 2000: 2% to 20% for soybeans, 4% to 11% for corn, and 2% to 8% for wheat.

- Estimates of total production losses for all three crops due to ozone are multiplied by the proportion of O_3 believed to be attributable to power plants to derive estimates of crop production benefits associated with the power plant development options examined in ORBES. If the assumptions of power plant NO_x contributions to regional O_3 are accepted, the utility-related portion of estimated soybean losses range from 8.3 to 72.1 million bushels in 1976, increasing to a range of 39.9 to

113.7 million bushels in 1985, and from 83.2 to 182.4 million bushels under the assumption of high electrical growth by the year 2000. Corresponding estimates for corn (in millions of bushels) range from 25.4 to 158.9 (1976), 44.5 to 190.7 (1985), and 85.3 to 281.8 (2000); for wheat, the estimates are 1.6 to 9.1 (1976), 2.7 to 16.4 (1985), and 6.4 to 29.4 (2000).

- These losses represent 1.5% to 33% of current production for soybeans, 1.2% to 13.3% for corn, and 0.9% to 16% for wheat. Within the limits of the data available, the projected losses for the three crops can be considered equivalent to potential crop production gains achievable from complete pollution abatement.

Possible

- A relatively untested null hypothesis regarding the time-trend of O_3 concentrations in the ORBES region would show present O_3 exposures as unchanged from historic levels, and, therefore, unlikely to be influenced by current or future interventions in the emissions of O_3 precursors. While this possibility cannot be excluded, it appears improbable at this time.

Section IX, considers the effects of SO_2 and O_3 on forest growth, mortality, and forest insect and pathogen interactions concluding with preliminary estimates of forest growth losses due to oxidants. The large number of native species and the greater duration of exposure for long-lived plants, relative to annual crops, complicates the analysis. In addition, the experimental data base on effects is much less extensive for forests than for crops, and the effects of primary and secondary pollutants (e.g. acid rain) cannot be separated. Findings include:

Factual

- Studies in both the western and eastern United States have documented reductions in annual ring width on conifers which have been attributed to air pollutants. Annual mortality rates are now 3% per year for ponderosa and Jeffrey pine in the San Bernardino Mountains.
- In the ORBES region, 37.3 x 10^6 acres of land are forested, with a regional average forest growth rate estimated to be 40 ft.^{3}/acre/year for all species. Approximately 4.2% of the 37.3 x 10^6 acres support coniferous species.

Probable

- Sensitive coniferous species are estimated to be experiencing reductions in wood accretion rates of between 0.04 and 0.18 ft.^{3}/tree/year, for the O_3 exposures being observed in the ORBES region. This leads to an estimated softwood growth reduction of between 2.76 to 7.33 million ft.^{3}/year for the entire ORBES region.

- Hardwood species are divided into three air pollution categories: sensitive, intermediate and resistant. Preliminary data suggest the sensitive hardwoods have the same reduction in growth rates as sensitive softwoods, for the same exposures, which calculates to an estimated reduction in growth rate of 3.36 to 17.9 million ft.^{3}/year. Intermediately resistant hardwoods make up a larger part of the annual hardwood growth and are estimated to be experiencing a reduction growth rate of one-half that of the sensitive species. The result is a reduction of 4.85 to 25.7 million ft.^{3}/year. Resistant hardwoods are not considered to be experiencing a reduction in growth rates at this time.

- The total estimated reduction in forest growth in the ORBES region due to air pollutants, principally O_3 is 10.97 to 50.93 million ft.^{3}/year. Normal annual growth for the ORBES region is estimated to be 1.49 billion ft.^{3}/year. Thus, the 1976 direct loss from air pollution stress on all species is estimated at this time to have been from 0.7 to 3.4 percent. Estimates of reduced growth due to mortality and insect or disease interactions cannot be made at this time.

- The estimated reduction in forest growth due to air pollutants in 1985 is from 1.3% to 6.3 % of the total production. For the year 2000 losses vary depending on energy development: with modest electrical growth, projected losses are estimated to be between 0.4% and 1.9%, and for higher energy growth, between 2.0% and 9.3 % of total production.

- Evidence exists which suggests insect damage is more prevalent in areas of air pollution comparable to that in portions of the ORBES region.

In an epilogue, Section X, the aggregate effects of ORBES region utility emissions are considered, including their potential contribution to acidity of rainfall outside the region and its ecological consequences there. ORBES area emissions undergo chemical transformations during transport beyond the region, and are deposited with enhanced hydrogen ion (H^+) concentrations (lower pH) in Canada and the northeast United States. The impacts of this acidity are difficult to estimate, and resource losses, such as the failure of lake fisheries, are poorly expressed as an annual loss of products. Findings include:

Factual

- Both historical and recent data on the pH of rainfall over the eastern United States indicates unusually high concentrations of H^+ (low pH) throughout the states generally downwind of the high-sulfur emitting areas of the ORBES Region.

- Acidic substances are known to be produced in the atmosphere when SO_2 and NO_x from fossil fuel combustion react to form sulfuric and nitric acids, as well as dry particulate sulfate and nitrate compounds. Meteorological studies show these pollutants can be transported (and deposited) hundreds of miles from their source.

- Recent reviews show that, despite some questions of rates of reactions and control mechanisms, enough is known about acidic deposition now to describe the general response relationships, the areas of impact, the mechanisms by which effects are expressed and the basis for lags in response times from one location to the next.

- Known effects of acid deposition on plants include partial removal of the waxy surfaces of leaves and mild lesions of leafy and root crops.

- Lakes in areas with granitic, quartzitic and sandstone bedrocks are especially sensitive to acid input due to their inherently low buffering capacity. Fish reproduction is disrupted below pH 5.0-5.5 in these areas and other aquatic organisms are adversely affected by similar and lower pH's.

Probable

- The direct effects of SO_2 emissions appear to be modest at the present time, and are likely to decrease under all scenarios. Indirect effects from SO_2 and NO_x emissions, including acid rain effects, are likely to increase (despite some decreases in total emissions) due to the longer lag-time in expression of watershed and aquatic effects.

- The effects from O_3 in the ORBES region, an apparent indirect result of NO_x emissions, are probably as large or larger than the SO_2 and acid rain effects combined. The trade-offs between sulfur oxides (SO_x) and nitrogen oxides (NO_x) emission controls and projected benefits appear to require a complex evaluation procedure.

- Although direct effects on plant foliage are unlikely at the present levels of precipitation pH, indirect effects (nutrient stripping and aluminum mobilization) could lead to decreased crop and forest productivity.

- The greatest effects of acid deposition on aquatic ecosystems appear to be expressed during spring snow melt period when accumulated H^+ (and mobilized aluminum) is flushed into receiving waters.

Possible

- Preliminary estimates indicate 100 to 200 additional lakes per year could be losing at least some of the reproduction of their game fishery. During the decade of the 1990's and beyond, this value seems likely to be increased, but ORBES region emissions may account for less than one-fourth of these effects. More complete analysis of the long-term resource effects and their value to society are urgently needed.

- If present trends toward lower forest productivity hold (Section IX), the reductions in forest growth could contribute to a "diminished CO_2 sequestering capacity" in the temperate zone forests, sufficient to be of significance to the global carbon balance, and CO_2 increases in the atmosphere.

SECTION III

A REVIEW OF MECHANISMS OF ACTION OF OZONE AND SULFUR DIOXIDE POLLUTANTS ON CROPS AND FORESTS

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INTRODUCTION

Numerous reviews have discussed the general relationships between plants and ozone (O_3) or sulfur dioxide (SO_2) over a range of concentrations and under a wide variety of light, water, soil, and varietal influences (Naegele 1973, Mudd and Kozlowski 1975, Heck and Brandt 1977). This review instead will focus on the mechanisms of action by these pollutants at concentrations that are as low as or lower than the current federal O_3 and SO_2 standards. The goal is to assess the potential for field effects on photosynthesis or plant growth that may occur under ambient exposures in the ORBES region, and possibly in the absence of visible damage.

Sulfur dioxide and O_3 concentrations in the ORBES region frequently have been reported in excess of the levels known to cause observable effects on certain crops and forest species. Recent monitoring shows the region to have a relatively high background oxidant level in rural areas during the summer (see later Sections).

This review of mechanims, therefore, will examine processes inhibited or otherwise altered by exposures at the concentrations being experienced in the Ohio River Basin, taking into consideration what is known about synergistic action between SO_2, O_3, and NO_x. Explanation of the processes leading to dysfunction of the normal physiological mechanisms of plants provides the foundation for evaluating the studies of yields effects on crop and forest species in later sections.

UPTAKE OF SO_2 AND O_3 BY PLANTS

The prerequisite for SO_2 or O_3 injury is entry of the gaseous pollutant into the plant. Species and varietal differences in resistance to SO_2 and O_3 may be explained by differences in the relative rate of absorption of the gas (Bressan et al. 1978), and thus in the diffusion density gradient. Once SO_2 or O_3 have found their way into a cellular environment, they may undergo transformations to yield other chemical compounds. The pollutants or their derivatives often initially inhibit many enzyme systems and metabolic processes. These biochemical changes accumulate to cause physiological disturbances such as reduction in photosynthesis translocation and growth.

Stomatal openings are the major route for gaseous pollutant uptake by plants. Gaseous flux into the leaf depends on molecular diffusion, governed by the pollutant concentration gradient from the leaf surface to the interior. The concept of leaf resistance, first developed to account for exchanges of oxygen, CO_2 and water vapor by plants, has been adapted to describe uptake of gaseous pollutants by plants. Pollutant uptake rates estimated by leaf resistance models are generally in good agreement with experimental data on total uptake (O'Dell et al. 1977).

Leaf resistance to gaseous flux can be thought of as consisting of four components: boundary layer resistance, stomatal resistance, cuticular resistance, and mesophyll resistance. Total leaf resistance is a combination of the four resistance components (Bennett et al. 1973, Unsworth et al. 1976). Since the magnitudes of molecular diffusive forces and leaf resistances can be estimated, net flux of the pollutant into a plant is determined by comparison of these opposing forces.

Control of stomatal "resistance" to uptake of gases is determined by the density and distribution of stomata and by whether of the apertures are open or closed. The concept of stomatal resistance is essential for understanding certain aspects of air pollution injury to plants. Because stomatal aperture is known to be affected by environmental factors such as light, temperature, humidity, and CO_2 concentrations, variations in plant response to air pollutants under different environmental conditions can be explained, at least in part, by the difference in stomatal resistance. Furthermore, disparate pollutant susceptibility both between and within plant species can be explained by stomatal openness. Runeckles and Rosen (1977) have shown that the tolerance of *Phaseolus vulgaris* L. to O_3 is due to stomatal closure following acute exposure to O_3. Other studies (Turner et al. 1972, Leone 1976) confirm that tolerance to air pollutants by some plants results, in part, from higher stomatal resistance.

The third resistance to flux of gases into a plant is cuticular resistance. The potential for $SO_2 \cdot O_3$ penetration through the cuticle is limited because the cuticle is a very effective barrier to gaseous penetration. There is no evidence to suggest that cuticular penetration contributes significantly to internal pollutant concentration. For all practical purposes, the flux of pollutants through the cuticle can be assumed to be zero.

The mesophyll resistance, a measure of the capacity of the leaf to remove SO_2 and O_3 molecules from the intercellular space, is a function of the solubility coefficient of the gases and the ability of internal leaf tissue to remove the pollutant from the site of uptake (Bennett et al. 1973). The extent of intercellular cell surface available for pollutant deposition, cell surface hydration, and rate of chemical transformation by the pollutant once it has dissolved in the aqueous media of the mesophyll cells, are three critical determinants of mesophyll resistance.

Physiological and structural aspects of the resistances to gaseous uptake by plants are critical to plant tolerance or sensitivity to injury by O_3, SO_2 and other air pollutants. Ashenden and Mansfield (1977) considered the importance of gas flux resistances when they increased wind velocities in their fumigation chambers and found increased biomass losses with very low doses of SO_2. They concluded that:

> "Where plants have been exposed to pollutants in virtually still air, the high boundary layer resistance reduces their rate of entry into the leaves, resulting in an overestimate of the concentration required to cause damage."

According to Ashenden and Mansfield, more normal (field) wind conditions would be expected to lower thresholds of damage, with or without symptom expression. Consequently, consideration of concentrations of pollutants and durations of exposure necessary to cause injury must include information on wind speeds in the fumigation chambers, if such equipment is used.

EFFECTS OF OZONE ON CROPS AND FORESTS

Several recent reviews have been published on the mechanisms by which O_3 operates on the biochemical, cellular, morphological and population responses of plants (Naegele 1973, Mudd and Kozlowski 1975, Mansfield 1976, U.S. EPA 1978, Montana Air Quality Bureau 1979). Most of the literature on the subject is contained in these reviews.

Certain trends are evident in this literature. The early work responded, in part, to observations made in the field by concerned farmers and agricultural scientists. When air pollution was first being evaluated as an agent of plant disease, O_3 concentrations much greater than ambient polluted levels were applied, and O_3 was found responsible for the plant maladies. The heavy doses utilized by the earlier workers caused perturbations to respiration, photosynthesis, nitrogen metabolism, growth, and flowering (Mansfield 1976). Later, researchers tended to reduce the concentrations of gases used to elicit injury and re-evaluated the knowledge of the mechanisms by which O_3 effects on vegetation are expressed. Some of the more recent results point to problems in using high concentrations of O_3 for short periods of time merely to elicit a change in a physiological process. Field fumigation studies now have been carried out at realistic concentrations as indicated by the results summarized in Table III-1.

Because O_3 is a natural constituent of the atmosphere, it is probable that organisms would have a tolerance to its presence at historic or natural background levels. Bennett _et_ _al_. (1974) suggest such a tolerance in their fumigation studies with beans, barley, and smartweed. However, when the concentration of O_3 is elevated to just 0.05 ppm for significant periods (levels which are present in the Ohio River Basin for

TABLE III-1. SUMMARY OF PAPERS REPORTING O_3-INDUCED EFFECTS ON SELECTED PLANTS AT LOW TO MODERATE CONCENTRATIONS. SOURCE: U.S. EPA 1978

Plant Species	Concentration	Exposure	Response of Vegetation	References
Alfalfa	0.05 ppm	7 hr./day, 68 days	30% leaf dry wt. reduction harvest 1 50% leaf dry wt. reduction harvest 2	Neeley *et al*. 1977
Alfalfa	0.10 ppm	2 hr./day, 21 days 2 hr./day, 21 days	16% leaf dry wt. reduction 25% leaf dry wt. reduction	Hoffman *et al*. 1975
Wheat, cultivar	0.20 ppm	4 hr./day, 7 days (anthesis)	30% yield loss	Shannon and Mulchi 1974
Soybean	0.05 ppm	8 hr./day, 5 day/wk. 21 days	13% leaf injury	Tingey *et al*. 1973
		8 hr./day, 5 day/wk. (mixture of O_3 & SO_2 for same period)	16% leaf injury 20% root dry wt. reduction	
Corn, sweet, cultivar-Golden Midget	0.05 ppm	6 hr./day, 64 days	9% kernel dry wt. reduction	Heagle *et al*. 1972
Bean, cultivar	0.13 ppm	8 hr./day, 28 days	79% leaf fresh wt. reduction 73% root fresh wt. reduction 70% height reduction	Manning *et al*. 1971
Bean, cultivar	0.15 ppm	2 hr./day, 63 days	33% biomass reduction 46% pod fresh wt. reduction	Hoffman *et al*. 1973
Bean, cultivar	0.15 ppm	2 hr./day, 14 days	8% leaf dry wt. reduction	Maas *et al*. 1973
	0.15 ppm	3 hr./day, 14 days	8% leaf dry wt. reduction	
	0.15 ppm	4 hr./day, 14 days	23% leaf dry wt. reduction	
	0.15 ppm	6 hr./day, 14 days	49% leaf dry wt. reduction	
Pinto beans	0.06 ppm	5 hr./day, 5 days/wk, 40 days	48% top wt. reduction 25% height reduction	Manning 1978

continued

TABLE III-1. CONTINUED

Plant Species	Concentration	Exposure	Response of Vegetation	References
Potatoes 4	0.05 ppm	730 days	34-50% yield loss	Heggestad 1973
	0.05 ppm	326-533 days	20-26% yield loss	
Potato cv.	0.15 ppm	3 days	95% leaf injury	Brasher et al. 1972
Radish	0.05 ppm	8 hr./day, 5 days/wk, 35 days	54% root fresh wt. reduction 20% leaf fresh wt. reduction	Tingey et al. 1971
	0.05 ppm	8 hr./day, 5 days/wk. (mixture of O_3 & SO_2)	63% root fresh wt. reduction	
Ponderosa pine	0.15 ppm	9 hr./day, 10 days	4% photosynthesis reduction	Miller et al. 1969
	0.15 ppm	9 hr./day, 20 days	25% photosynthesis reduction	
Ponderosa pine	0.15 ppm	9 hr./day, 30 days	25% photosynthesis reduction	
	0.15 ppm	9 hr./day, 60 days	34% photosynthesis reduction	
Ponderosa pine	0.10 ppm	6 hr./day, 126 days	12% root length reduction 21% stem dry wt. reduction 26% root dry wt. reduction	Wilhour & Neeley 1977
Western pine	0.10 ppm	6 hr./day, 126 days	13% leaf dry wt. reduction 9% stem dry wt. reduction	
Poplar, hybrid	0.15 ppm	8 hr./day, 5 days/wk, 42 days	50% shoot dry wt. reduction 56% leaf dry wt. reduction 47% root dry wt. reduction	Jensen & Dochinger 1974
14 plants from an Aspen forest	0.15-0.30 ppm	3 hr./day, 5 days/wk. through the growing season	significant amounts of leaf injury and losses in plant wt. and seed wt.	Haward & Treshow 1975
Petunia	0.05-0.07 ppm	24 hr./day, 53 days	30% flower fresh wt. reduction	Craker 1972
Carnation	0.05-0.09 ppm	24 hr./day, 90 days	50% reduced flowering (reduced growth)	Feder 1970

much of each growing season, see Figure 4 of Section V), several key plant species may be affected to the point of potential economic crop failure (see Table III-1 for alfalfa, soybeans, sweet corn, pinto beans, potatoes, radishes, petunias, and carnations).

A natural resistance to O_3 and other air pollutants has been developed for some species during selection trials for plant resistance or tolerance. However, the cumulative dose functions observed on ponderosa pine in southern California (Miller et al. 1978) may be at variance with ideas about permanent O_3 tolerance. Ponderosa pines were tagged early in the 1950's and 1960's when they showed apparent resistance to oxidant pollutants, i.e., showed no obvious disease symptoms. As time passed, however, and the annual dose increased over several decades, the trees thought originally to be resistant to oxidants became diseased.

This observation has implications for understanding the biochemical and genetic pathways by which O_3 operates on living plant systems. A widely held view is that O_3 first affects photosynthesis, bleaching the chlorophylls and reducing carbohydrate manufacture, resulting in reduced growth. The plant organs furthest away from the leaves appear to be affected the most, suggesting that translocation is one of the most O_3-sensitive processes. Inhibition of translocation would be a secondary response due to membrane disruptions, enzyme coagulation, reduced photosynthates, osmotic dislocations, and lowered ATP production (Mansfield 1976). All of these processes have been demonstrated to be adversely affected by O_3 and related oxidants.

Since photosynthesis and translocation are inhibited by O_3, cellular growth, leaf, flower, and root growth inhibitions can result (Table III-1). Generally, O_3 seems to operate so as to accelerate the aging process. Catabolic physiological processes, such as leaf abscission, appear to be stimulated. Anabolic physiological processes, such as photosynthesis, are inhibited or slowed down. In some plants, e.g., beans, it appears that enzymes are mobilized or activated to repair O_3 damage (Table III-2). Reducing sugars in leaves may be increased because of inhibition of translocation from mesophyll areas. Increased formation of pigment, such as anthocyanin may result, a mechanism toughening tissues against further oxidation by O_3 similar to resistance mechanisms against fungi (Mansfield 1976).

Horsman and Wellburn (in Mansfield 1976) have listed metabolic processes and biochemical reactions affected by O_3 (Table III-2 and Table III-3). Several key biochemical components in the citric acid cycle are increased in response to O_3. The increases may be due to an accumulation of metabolites as respiration is inhibited further around the cycle, or because of O_3 induced increases in respiration rate. Stimulated respiration as an O_3-induced response would be expected to increase the organic acid constituents of the Kreb's citric acid cycle and to increase ATP as mitochondrial membranes are disrupted, which prevents electron shunt and transfer from ADP to ATP and back again. Ozone-damaged cells also in-

TABLE III-2. METABOLIC AND BIOCHEMICAL FACTORS INCREASED BY OZONE CONCENTRATIONS AT OR BELOW 0.15 ppm FOR VARIOUS EXPOSURE PERIODS*

Species	Factor Increased
Beans	Sterol glucoside Esterified sterol glucoside Disulfide level Peroxidase Ribonuclease
Pines	Soluble sugar level Ascorbic acid level Total soluble sugar level
Corn and soybeans	Nitrate reductase (higher at 0.062 ppm/4 hr; lower at 0.05 ppm/2 hr)
Peas	Cell membrane permeability (changed)
Spinach	Formic acid

*After Horsman and Wellburn, in Mansfield (1976)

TABLE III-3. METABOLIC AND BIOCHEMICAL FACTORS DECREASED BY OZONE CONCENTRATIONS AT OR BELOW 0.15 ppm FOR VARIOUS EXPOSURE PERIODS*

Species	Factor Decreased
Beans	RNA level Lactate dehydrogenase enzyme Starch hydrolysis in the dark Chlorophyll level Free sterol Monogalactyl diglyceride
Pines	Polysaccharide level
Soybeans	Chlorophyll level Total soluble sugar level Reducing sugar level Nitrate reductase NADPH-dependent nitrate reductase
Cucumis and Cassia	Starch hydrolysis
Petunia	Ascorbic acid level
Monkey flower	Starch hydrolysis in the dark
Corn	Chlorophyll level Alkaline pyrophosphatase enzyme

*After Horsman and Wellburn, in Mansfield 1976 crease use of those biochemical building blocks necessary for cellular repair (Horsman and Wellburn, in Mansfield 1976).

Some cellular substances such as nitrate reductase enzymes have shown an increase/acceleration and decrease/inhibition to O_3 exposure, where a low dose increases the activity, and a higher O_3 dose lowers activity. Hormonal changes have been proposed that might initially result in temporary growth increases but, on an accumulated dose basis, would cause reduced growth. Tingey et al. (1976) and others have related the growth stimulation to production of ethylene, a growth hormone produced by stress mechanisms in the presence of O_3.

The metabolic dysfunctions observed are believed to be associated with ultrastructural disruptions induced by O_3 (Matsushima et al. 1977). Chloroplast lamallae, deteriorated ribosomes and polysomes, increased vacuole size, plus slight plasmolysis and small breaks in membranes, all would be associated with cellular leakage and increased sugar intake as stressed and injured plants attempt to repair cells and tissues damaged by the oxidizing activity of O_3.

Chronic exposure of plants to low dosages of O_3, plus a few acute episodes (enough to cause some visible injury), forces cells and tissues to become disjunct in their homeostasic mechanisms. After doses accumulate (dose = concentration x time*), the cellular metabolites and chemical reserves normally used for growth and regeneration are depleted, thus, growth and yield, as well as crop quality, are inhibited. Miller et al. (1977) showed that an accumulated O_3 dosage of 2.6 to 3.0×10^5 micrograms per cubic meter over a 5-month growing season from May through September (avg. 0.06 ppm O_3) would produce chlorotic mottle on current year needles of ponderosa pine under natural conditions. However, by the time current needles become visibly diseased, older needles may already have become necrotic or much more diseased because of acute episodes (Williams 1980). These results are consistent with the hypothesis that senescence is accelerated by O_3.

The normal partitioning of solutes to roots, shoots, and fruits also is inhibited or slowed down by O_3, and can eventually be felt as an economic loss. Effects on metabolism appear to function so as to reallocate photosynthates and other metabolites away from biomass accumulation sites during growth, and direct more of the biochemical components into cellular or tissue maintenance according to the work of Thornley (1970). Bennett et al. (1976, 1979) and Oshima et al. (1977a) exposed several important vegetable crops to low chronic doses of O_3 and observed dysfunctions in partitioning abilities of the test plants, which caused lower yields.

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* A further refinement for "dose" would be to include a factor for frequencies of acute episodes.

They explained that plant systems have a wide variety of alternatives for providing a continual response to stress during exposure to pollutants.

Application of low concentrations of O_3 over extended periods of time (at least as long as annual growth cycles) has led to new views of the mode of action of O_3 in vegetation, principally that ozone acts in a cumulative dose-response manner (Oshima et al. 1976, 1977b, Miller et al. 1978, Williams 1980). Injury to cellular material can accumulate over long periods of time from relatively low concentrations of O_3, e.g., just above oceanic clean air ambiency of 0.03-0.04 parts per million (ppm). The period of exposure and the kinds of episodes determine the severity of the effects expressed. For example, a low chronic dose of 0.05 ppm by itself can cause significant crop effects, but associated acute episodes of 0.08-0.10 ppm or higher augment the injury (Table III-1). Thus, with the increasing evidence of cumulative dose-response, long-term exposures to low oxidant levels are believed to result in effects as severe as the injury caused by higher but short-term exposures.

On a regional economic basis, accelerated senescence is probably a most critical mode of action of O_3. Since the timing of harvest is fine-tuned for most agricultural crops, delayed or accelerated ripening by only a few days may mean reduced crop values. Premature leaf abscission, resulting from either chronic O_3 exposure or the shock of an individual episode, reduces the esthetic value of crops such as flowers, and of crops up for auction (tobacco, alfalfa, other hay crops). Although crop biomass (yield) may not be affected, impaired appearance of these and crops such as Christmas trees, renders them of limited sale value.

EFFECTS OF SULFUR DIOXIDE ON CROPS AND FORESTS

The research literature on SO_2 effects on vegetation has been reviewed numerous times in recent years. As with O_3, early workers fumigated test plants with large doses of SO_2 to show that SO_2 could cause damage. However, SO_2 exposures came largely from point sources--metal smelters, coal-fired power plants and other industries--instead of from area-wide sources as with O_3. In many instances devastation of vegetation occurred creating gradients of damage. These zones became natural laboratories. Examples of such sites include the areas around Sudbury, Ontario; Biersdorf, Germany; Trail, British Columbia; and Anaconda, Montana. In each case, the most sensitive species disappeared, and in some cases (but not always) they were succeeded by SO_2 tolerant species. Following abatement, the areas have recovered slowly, and long-term downwind economic losses have been estimated (e.g., Linzon 1972). The extensive experimental work carried out to evaluate these observed effects is summarized in Table III-4 with an emphasis on studies using some treatments at relatively low concentrations and long exposures.

Biochemical Effects

Following absorbtion of SO_2 by plants, biochemical interactions occur between the absorbed pollutant molecules and constituents in the cells. Exposure of plants to radioactive SO_2 ($^{35}SO_2$) shows the products to include sulfite, sulfate, cysteine, glutathione, and methionine (Weigal and Ziegler 1962, Garsed and Read 1977a, 1977b).

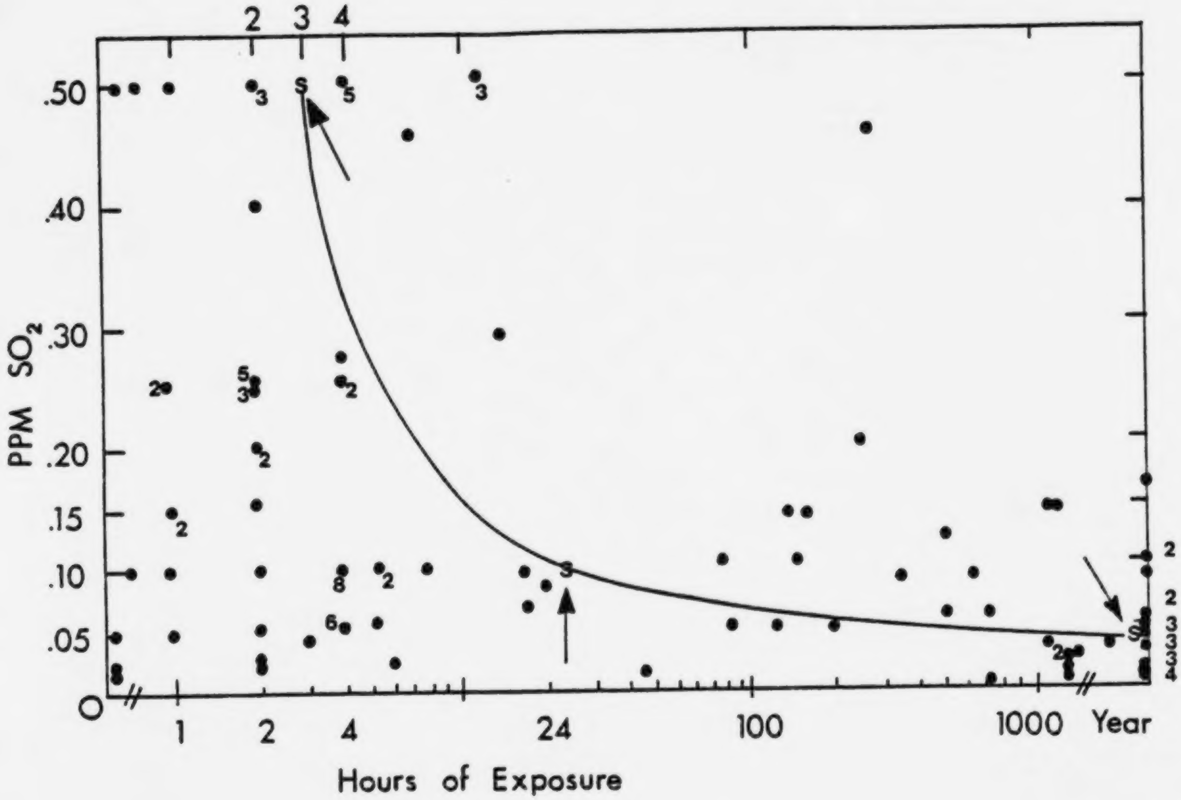

Figure III-1. Comparison of federal ambient air quality standards (arrows) for SO_2 with SO_2 doses that produced significant plant and ecosystem stress. Each dose point in the figure corresponds to a literature citation in Table 4. For example, at 0.50 ppm SO_2 for a 4-hour exposure, five experiments were completed. Alfalfa at that dose suffered 19% foliar injury (Tingey _et al_. 1973). At 0.10 ppm SO_2 for 4 hours, eight experiments were completed. Alfalfa with 0.10 ppm O_3 plus 0.10 ppm SO_2 suffered 24% leaf injury (Tingey _et_ _al_. 1973).

TABLE III-4. SULFUR DIOXIDE EFFECTS--ALONE AND IN COMBINATION WITH OTHER POLLUTANTS--TO SELECTED PLANTS AT OR BELOW 0.30 ppm*

Plant Species	Concentration	Exposure	Response of Vegetation	References
Grain Crops				
Corn at 4-leaf state	0.01 ppm SO_2	10 min. 20 min.	Stomata opened wider (measured as decreased diffusive resistance).	Unsworth et al. 1972
Oats, barley, alfalfa	0.2 ppm SO_2	2 hours	Photosynthesis inhibition begins (as measured by CO_2 uptake rate compared to controls).	Bennett and Hill 1974
Wheat	0.015 ppm SO_2	Growing season	15% reduction in grain wt. yield	Guderian 1977
Oats	0.01 ppm SO_2	Growing season	Leaf injury	Guderian 1977
Oats	0.05 ppm SO_2 + 0.05 ppm NO_2	4 hours	Tissue death and loss of chlorophyll	Tingey et al. 1971
Legumes				
Alfalfa	0.15 ppm SO_2 + 0.15 ppm NO_x	1 hour	Inhibits photosynthesis; percent reduction not stated, but authors state synergistic effect most marked at this concentration.	White et al. 1974
Alfalfa	0.15 ppm SO_2 + 0.15 ppm NO_2	1 hour	Greater-than-additive inhibition (7%) of photosynthesis (as measured by CO_2 uptake). Similar results at 0.25 ppm for both gases.	Bennett and Hill 1974
Alfalfa	0.30 ppm SO_2 + 0.10 ppm Ozone (O_3)	1 hour	Additive inhibition of photosynthesis (11% ± 3% measured by CO_2 uptake). Some tissue damage.	Bennett and Hill 1974
Alfalfa	0.15 ppm SO_2 + 0.15 ppm NO_2	2 hours	7% reduction in apparent photosynthesis. Some tissue death.	White et al. 1974.
Alfalfa	0.10 ppm SO_2 + 0.10 ppm O_3	4 hours	24% leaf injury	Tingey et al. 1973

(continued)

TABLE III-4. CONTINUED

Beans at 4-pair and 5-pair leaf stages	0.05 ppm SO_2	10 min.	Stomata opened wider (measured as decreased diffusive resistance); similar response at 0.10 ppm and 0.45 ppm. Oldest leaves most sensitive (stomatal resistance increased with leaf age irrespective of SO_2 concentrations).	Biscoe et al. 1973
Beans	0.02 ppm SO_2	10 min.	Stomata opened wider (measured as decreased diffusive resistance).	Unsworth et al. 1972
Beans at 4-leaf stage	0.10 ppm SO_2	1 hour	Stomata opened wider (measured as decreased diffusive resistance).	Unsworth et al. 1972
Spongy mesophyll of 25-day-old broad bean leaves at second node from base	0.25 ppm SO_2	1 hour	Slight swelling of stroma thylakoids of chloroplasts. Author states implication is impairment of photosynthesis. At 2 hrs. at 0.25 ppm swelling spread to granum thylakoids. These effects reversible.	Wellburn et al. 1972
Pinto beans	0.05 ppm SO_2 + 0.05 ppm NO_2	4 hours	2% leaf injury measured as pigmented lesion, tissue death, loss of chlorophyll.	Tingey et al. 1971
Beans and corn	0.10 ppm SO_2	20 min.	Stomata opened wider (measured as decreased diffusive resistance). No difference between plants exposed in dark or light: SO_2 opens stomates in the dark.	Unsworth et al. 1972
Peas	0.05 ppm SO_2 + 0.05 ppm NO_2	1 hour	Significantly decreased net photosynthesis	Bull and Mansfield 1974
Peas	0.10 ppm SO_2 + 0.10 ppm O_3	5 hours	14.6% damage. Leaf surface destruction more severe on lower leaves. Ash-colored spots on intermediate leaves and veins.	Fujiwara et al. 1973
Peas	0.05 ppm SO_2 + 0.10 ppm O_3	5 hours	9% visible injury. Upper leaves showed ozone injury; lower leaves showed O_3 leaf surface destruction. Author calls this response synergy.	Fujiwara et al. 1973

(continued)

TABLE III-4. CONTINUED

Radish	0.10 ppm SO_2 + 0.10 ppm O_3	4 hours	50% leaf injury.	Tingey et al. 1973
Radish	0.10 ppm SO_2 + 0.10 ppm O_3	4 hours	50% leaf injury.	Tingey et al. 1973
Tomato	0.10 ppm SO_2 + 0.10 ppm O_3	4 hours	50% leaf injury.	Tingey et al. 1973
Tomato	0.20 ppm SO_2	15 days	First sign of tissue death. By this time the vitamin content has been substantially reduced or changed. (B_1, B_6 Nicotinic acid.)	Unzicker et al. 1975
Forest Species and Trees				
Eastern white pine	0.05 ppm SO_2	2 hours	Caused tip damage on new needles.	Costonis 1973
Eastern white pine	0.25 ppm SO_2	2 hours	6.5% of exposed parts injured. Needle damage.	Berry 1971
Eastern white pine	0.05 ppm SO_2 + 0.05 ppm O_3	4 hours	Severe needle injury (2-3cm leaf tip injured) on new needles, expressed most vividly 72 hours after exposure.	Costonis 1973
Eastern white pine	0.025 ppm SO_2	6 hours	Threshold for needle damage. Response judged in terms of needle elongation (growth) and needle lesions or tip damage. 0.025 SO_2 + 0.05 O_3 showed significant response to all sensitive plants.	Houston 1974
Jack pine	0.25 ppm SO_2	2 hours	4.5% of exposed parts injured. Needle damage.	Berry 1971
Jack pine	0.045 ppm	Growing season	Injury to Jack pine trees was a mixture of chlorophyll loss and tissue death.	Linzon 1978

(continued)

TABLE III-4. CONTINUED

Scotch pine, Norway spruce, Douglas fir, Frazier fir	less than 0.01 ppm SO_2 0.01 ppm NO_2 0.07 ppm HF 0.05 ppm O_3 four month average during growing season	Growing season	Growth abnormalities, needle loss and tissue death.	EPA 1971
White birch	0.006 ppm	Growing season	No effects on sulfur content of foliage development of leaf SO_2 injury.	Linzon 1978
	0.011 ppm	Growing season	Elevated sulfur levels trace to light leaf SO_2 injury.	
	0.025 ppm	Growing season	Moderate to severe leaf SO_2 injury.	
	0.048 ppm	Growing season	Severe SO_2 injury symptoms and leaf sulfur concentrations 3 times normal.	
Cottonwood	0.3 ppm SO_2	4 hours	Significant decreases in moist pollen tube length.	Karnowsky et al. 1974
Lodgepole pine	100 to 500 ppm aqueous = 0.1 to 0.5 ppm in the air	22 hours	Disruption of chloroplasts and reduction in photosynthesis	Malhotra 1976
Douglas fir seedlings	0.29 ppm SO_2	44 hours	Visible injury.	Katz and McCallum 1939
Loblolly pine (2 wks. old), Sycamore (1 wk. old), seedlings	0.14 ppm SO_2 + 0.05 ppm O_3 + 0.10 ppm NO_2	6 hours/day for 28 days (168 hours)	Significant growth reduction (measured as height) compared to either O_3 and SO_2 combined, or O_3 alone. Needles were significantly shorter than for any other exposure. This study is an example of growth reduction with slight leaf symptoms. Leaf symptoms response most sensitive in early July.	Kress and Skelly 1977
Birch and Aspen foliage	0.008 ppm	Growing season	Very slight injury 64 km NE of SO_2 source.	

(continued)

TABLE III-4. CONTINUED

Fir forests	0.007 - 0.01 ppm	Annual average	20 ± 5% loss in growth. Premature drop of some needles, and reduction of above-growth plant organs.	Materna 1973
Fir forests	0.026 - 0.037 ppm	Annual average	Rapdi death of whole groups of trees.	
Grass and Range Species				
Orchard grass	0.11 ppm SO_2	4 weeks	34% reduction in leaf size, 45% reduction in dry weight.	Ashenden 1978
Ryegrass	0.11 ppm SO_2	4 weeks	20% reduction in leaf size, 40% reduction in total dry weight.	Ashenden and Mansfield 1977
Ryegrass	0.12 ppm SO_2	9 weeks (504 hours) 26 weeks (1456 hours)	46% reduction in yield measured as leaf and tiller production; increased loss of leaves, 52% reduction in dry shoot weight.	Bell and Clough 1973
Ryegrass	0.073	26 weeks (4368 hours)	50% reduction in dry weight of shoots.	Bell amd Mudd 1976**
Ryegrass	0.37 - 0.19	Periodic harvest between 6 and 43 wks.	Whole plant dry weight of ryegrass grown either on a nutrient-rich sand culture or on unfertilized field soil was depressed for up to 36%.	Crittenden and Read 1978,1979**
A number of Montana range	SO_2 concentions ranging from 0.02 ppm, 0.05 ppm, 0.15 ppm, 0.25 ppm	Growing season	Few responses during 1st year of exposure. A number of subtle effects at 2nd year including: Earlier leaf fall, insect exodus, reduced species diversity, root/shoot impairments, possible seed germination delay, inhibition of litter decomposition.	U.S. EPA Colstrip 1979
Ricegrass	0.06 ppm SO_2	6 weeks	Reductions in photosynthesis, respiration, and the chlorophyll content. Slight increase in productivity.	Ferenbaugh 1978
Ricegrass	0.13 ppm SO_2	6 weeks	Decrease in productivity.	

(continued)

TABLE III-4. CONTINUED

Aquatic and Lower Plants				
Duckweed	0.15 ppm SO_2	6 weeks (1008 hours)	Reduction in size of fronds; starch content reduced.	Fankhauser et al. 1976
Duckweed	0.15 ppm SO_2	51 days (1224 hours)	Reduction in starch, reduction in growth (measured as reduced surface area, average dry weight, reduced doubling time).	Fankhauser et al. 1976
Lichens	0.041 ppm plus dust at 5 g/m^3 ambient	Monthly	Lichens completely absent.	Kirschbaum 1971
Lichens	0.042 - 0.030	6 months	Tissue death.	LeBlanc et al. 1972
Lichens	0.006 - 0.018	6 months	Loss chlorophyll and growth reduction.	LeBlanc and Rao 1973
Lichens	0.01 - 0.02	Annual	Elimination of many lichen species.	LeBlanc et al. 1972
Lichens	Below 0.011 ppm	Long-term	No effect.	Hawksworth 1973
Bryophytes	0.17	6 months (winter)	Elimination of plants.	Gilbert 1968
Medicinal Plants and Ornamentals				
Bel B. Tobacco	0.25 ppm SO_2	4 hours	1% leaf injury	Tingey et al. 1973
Tobacco	0.10 ppm SO_2	8 hours	Threshold for fleck-like lesions on leaves.	MacDowall and Cole 1971
Tradescantia	0.074 ppm SO_2	18 hours	Increased chromosone aberration rate of germinating pollen.	Ma et al. 1973

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* All bibliographic references from Montana Ambient Air Quality Standards Study, 1979-80.

** For supporting sequels see New Phytologist, 1979, Vol. 83, No. 3.

Sulfite and its dissociation products are considered largely responsible for the physiological damage caused by SO_2 (Puckett et al. 1973). Sulfite reacts with at least five groups of cytoplasmic compounds -- proteins, enzymes, aldehydes, ketones, and olefins. In particular, sulfite reacts with disulfide bonds in polypeptide chains with the production of thiol and S-sulfonate:

$$RSSR + SO_3^{2-} \rightleftarrows RS^- + RSSO_3^-$$

In addition to attacking the S-S bonds of an enzyme molecule (a protein), sulfite can inactivate enzymes by competitive inhibition of reactive sites. Ziegler (1972, 1973) showed that the inactivation of ribulose-diphosphate carboxylase and phosphoenolpyruvate carboxylase is due to competitive inhibition for the bicarbonate site.

The reaction of sulfite with aldehydes and ketones results in the formation of sulfonate:

$$R_1(R_3)CO + HSO_3^- \rightleftarrows R_1(R_2)C(OH)SO_3^-.$$

Mudd (1975) suggested that these reactions can affect biological systems in at least two ways: (1) trapping metabolic intermediates that have aldehyde or ketone groups, and (2) forming α-hydroxysulfonates that inhibit enzyme catalyzed reactions. Sulfite also reacts with olefinic compounds to produce sulfonic acids. For example, sulfite reacts with maleic or fumaric acids to produce sulfocarboxylic acids (Mudd 1968). The biological significance of some of these reactions is not fully understood.

Physiological Disturbances

Biochemical reactions of SO_2 result in some important physiological disturbances such as plasmolysis, photosynthesis inhibition, growth rate depression, and decreased buffering capacity (Table III-4). The degree of disturbance appears to depend primarily on the amount of pollutant absorbed.

Acute injury results from the rapid absorption of large quantities of SO_2. The cell membrane is normally the first structure to show effects. Disruption of the cell membrane is caused by an attack on sulfhydryl and lipid linkages, leading to a loss of the differential permeability of the membrane (Chimiklis and Heath 1975). Severe damage to cell membranes causes loss of cellular water and solutes, following which the cell plasmolyzes and collapses.

Chronic injury to the cell results from exposure to a concentration lower than the level that would cause cell rupture and biochemical reactions. Photosynthesis inhibition and growth rate depression, both physiologic consequences of low-level SO_2 uptake require special consideration. As with O_3, disruption of these processes may cause accelerated senescence.

The effect of SO_2 on chlorophyll has been known for some time. Rao and LeBlanc (1966) showed that exposure of the lichen *Xanthonia fallax* to SO_2 caused the conversion of chlorophyll to pheophytin, and Puckett *et al*. (1973) reported oxidation of chlorophyll as a result of SO_2 absorption by plants. In a histological study, Wellburn *et al*. (1972) found that exposure to 0.25 ppm for 1 hour caused swelling of broad bean chloroplast stroma and that high doses caused serious damage to chloroplast structure. Similar results were reported by Malhotra (1976) in chloroplasts of pine needles exposed to SO_2. These chemical and structural changes may be the principal reasons for the observed reduction of net photosynthesis by plants exposed to SO_2 (Sij and Swanson 1974, Silvius *et al*. 1975). One recent study (Lamoreaux and Chaney 1978) showed that net photosynthesis was reduced by 50% in excised leaves of silver maple exposed to 1 ppm SO_2 for 30 minutes. Other causes of plant growth reduction by SO_2 include decreased assimilative capacity, enhanced leaf senescence, and increased respiration.

Implications of the effects of SO_2 in the biochemical and cellular systems need to be extrapolated to whole plants and field populations. The recent ambient fumigation experiments of Crittenden and Read (1979) and Bell *et al*. (1979) at very low levels of SO_2 (0.0159 ppm-0.018 ppm-0.033 ppm) for 56-194 days on pasture grasses resulted in significant reductions in dry weight on the treated sites. These concentrations would not have been expected to elicit injury if viewed only from the short-term fumigation basis of a few hours exposure. Likewise, the results at Colstrip, Montana (U.S. EPA 1979), from low to moderate SO_2 fumigation regimes (0.02, 0.04, 0.07 ppm) during the growing season pointed to reduced sexual reproduction of grasses, decreased mycorrhizal populations, exodus of predator beetles, protein reduction on dominant grass species, accelerated foliar senescence, and negative effects on lichens.

Regional Implications

Of greater interest to the Ohio River Basin, however, are downwind areas relatively remote from high impact zones, where ecosystem species diversity has been reported to be affected, as well as ecosystem productivity, from annual concentrations as low as 0.01 ppm SO_2 (Table III-4). Laboratory, greenhouse, and field fumigation studies over extended periods are repeating the findings for both SO_2 and O_3. As monitoring instruments and fumigation techniques have improved, results are tending to show that SO_2 is more toxic at lower concentrations than previously believed.

Evidence supporting the high toxicity of SO_2 to vegetation is listed and described in Table III-4 and Figure III-1. In Figure III-1, the available experimental results are shown in relation to the current federal ambient air quality standards for SO_2. In the majority of fumigation experiments, especially when SO_2 was used in combination with O_3, nitrogen dioxide, and/or hydrogen fluoride, the federal standards for SO_2 did not protect vegetation. Although effects are especially apparent when SO_2 and other gases are used in combination, damage at exposures below the federal standard are not limited to mixed gas regimes.

The concept of effects of SO_2 acting in conjunction with other gases to cause additive and synergistic effects has been reviewed by Williams (1978), Linzon (1978), and Heagle and Johnston (1979). Synergistic effects between SO_2 and other gases, especially NO_2 and O_3, appear to be the rule at the lower concentrations of gases commonly observed in polluted environments. Only when concentrations of gases higher than those commonly encountered in nature are utilized in fumigation experiments does a quenching effect occur. High-concentration fumigation experiments, apparently, do not describe lower concentration plant response processes, but are of some value in showing end-result effects of chronic accumulation on biochemical and physiological processes.

CONCLUSIONS

The major entry route of SO_2 and O_3 into plants is via stomatal openings. The amount of pollutant uptake is influenced by the various plant canopy resistances. Boundary layer resistance is influenced by wind flow while mesophyll resistance depends on the solubility coefficient of gases and the ability of internal leaf tissues to remove pollutants. The stomatal aperture is affected by environmental factors such as light, temperature, humidity, and CO_2 concentrations. Once taken up by a plant, SO_2 and O_3 can cause biochemical reactions that in turn produce physiological disturbances manifested eventually as lower crop productivity and quality.

Long-term exposures to even moderately low oxidant levels eventually can result in effects as severe as the injury caused by higher short-term exposures. Synergistic reactions between SO_2, O_3, and NO_2 are well-documented, mostly increasing the damage done to plants by the individual pollutants. Whether plants can develop a true tolerance to O_3 and SO_2 appears to be in doubt due to the fact that pollutant-induced physiological disruptions, including disruption of translocation, may speed up the aging process. Such accelerated senescence can effect crop values by altering the critical time of harvest, causing premature loss of leaves, and damaging the esthetic appearance or quality of certain crops.

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SECTION IV

ASSESSMENT OF AIR POLLUTION INJURY TO EASTERN WHITE PINE IN INDIANA

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INTRODUCTION

To assess regional aspects of air pollution damage on vegetation, the principal goal of the TIE support study, some direct field observations were essential. Therefore, a field assessment was made of air pollution symptoms on eastern white pine (*Pinus strobus* L.) at nine stands in central and southern Indiana and four stands in Wisconsin.

Previous studies conducted in the field and in controlled environmental growth chambers have shown that eastern white pine is sensitive to air pollutants at presently occurring concentrations (Berry and Hepting 1964, Blanchard *et al*. 1979, Costonis 1970, Davis and Wood 1972, Dochinger and Seliskar 1970, Gerhold 1977, Houston and Dochinger 1977, Skelly *et al*. 1972). Ozone (O_3) and sulfur dioxide (SO_2) are the two pollutants that cause the greatest expression of injury symptoms on a large scale. Eastern white pine was selected for this study, therefore, because of its known pollutant sensitivity, comparative abundance throughout the ORBES region (native and widely planted), and easily accessible foliage. Stands were selected in areas with suspected differences in pollutant exposures, to differentiate symptoms associated with (1) rural concentrations of O_3 and/or other local air pollutants, (2) point source emissions of SO_2 from neighboring large coal-fired power plants (in conjunction with rural O_3 concentrations), and (3) air pollution in the Indianapolis urban area. Selection of sites was based on the number of trees in a stand, location with respect to point sources, exposure to pollutants, access to foliage, and remoteness from the effects of winter salt applications on roads.

Background Data: Ozone

Ozone is the triplet oxygen molecule, O_3, formed when hydrocarbons and nitrogen oxides react in the presence of light. Natural sources of O_3 in the lower atmosphere are considered minor, compared with anthropogenic sources (Applied Science Associates 1978). O_3 precursors are emitted from motor vehicles as well as fossil-fueled power-generating stations, and large urban areas usually are the major source of such precursors. Thus, the highest concentrations of O_3 often are found downwind of major urban centers (Jacobson 1979, Unger unpublished, White *et al*. 1976). Elevated concentrations of O_3 also have been documented in the plume of a coal-fired power plant (Miller *et al*. 1978).

Ozone is thought to cause more damage to vegetation in the United States than any other air pollutant (Applied Science Associates 1978, Gerhold 1977), injuring a wide variety of plants in all areas of the country. One reason for this is that the highest concentrations of O_3 are correlated with that part of the day when plants are highly active photosynthetically and therefore most susceptible to air pollutants. The daily oscillation in O_3 concentration follows a characteristic pattern, with low concentrations at night, a rising concentration during the morning hours, a peak in the afternoon, and a recession in concentration during the night.

The effects of O_3 on coniferous trees may be acute or chronic. After an acute exposure, conifers typically develop tipburn (tip necrosis), flecklike necrotic lesions, tissue collapse, and, in less severe cases, chlorotic mottle. Premature defoliation also may occur, but this is generally considered a symptom of chronic exposures (U.S. Forest Service 1973, Applied Science Associates 1978, Costonis 1968 and 1970, and Taylor 1969). According to Costonis (1968, 1970) and Gerhold (1977), acute symptoms first appear as silver flecks, which then change to yellow-pink spots later becoming necrotic bands and tip necrosis.

Chronic exposure to O_3 may produce chlorotic mottle, pigment changes, premature defoliation, tipburn and reduced growth rate. Other symptoms include reduction in terminal and diameter growth, decrease in the number and size of needles, deterioration of the fibrous root system, gradual decline and sometimes death of the tree (Applied Science Associates 1978, Blanchard _et al_. 1979, Costonis 1968, Taylor 1969).

O_3 decreases the ability of plants to withstand severe environmental conditions such as drought, insects, and fungal infections. It may contribute, along with SO_2, to a decrease in cone production. Houston and Dochinger (1977) found that white pines chronically exposed to air pollution in central Ohio showed significant decreases in the number of seeds per cone, 100-seed weight, and percent pollen germination; none of the trees showed foliar symptoms of air pollution. O_3 may also affect physiological processes without visible injury other than reduction in growth. Photosynthesis was decreased, and respiration increased, in seedlings of eastern white pines when they were exposed to 5 to 15 parts per hundred million (pphm) O_3 for 5 to 18 weeks (Gerhold 1977).

Gerhold (1977) noted that acute injuries to emerging foliage of white pine may occur from doses of O_3 as low as 3 pphm for 48 hours or 7 pphm for 4 hours. Fumigation with 10 pphm for 2 hours caused emergence tipburn. That these concentrations are damaging to the foliage of eastern white pine is supported by the work of Blanchard _et al_. (1979) in New Hampshire. Monitored O_3 concentrations exceeded 3 pphm on two occasions for 56 and 62 consecutive hours, with levels above 7 pphm for 4 and 7 consecutive hours during those periods. Both episodes were followed within 10 days by increases in severe needle damage to sensitive trees. Genetically identical material (scions) did not show such symptoms in areas with lower O_3 concentrations. Similar episodes and effects were recorded in 1974, 1975, and 1976; for example, 39 consecutive hours above

3 pphm O_3 produced noticeable needle damage in July, 1975. All these studies were carried out under field conditions, with trees exposed only to ambient levels of air pollutants.

Davis and Wood (1972) exposed 2- to 6-year-old seedlings of 18 coniferous species to either 10 pphm O_3 for 8 hours or 25 pphm O_3 for 4 to 8 hours and found chlorotic mottle and tip necrosis or entire needle necrosis. The threshold dose for visible injury to eastern white pines has varied from 3 to 10 pphm for 1 to 48 hours, with moderate doses ranging from 10 to 25 pphm for 48 hours. Such variation may be due to genetic variation among test plants, environmental differences in the individual experiments, or seasonal variation. Concentrations as high as 21 pphm have been recorded in Indianapolis, where values between 4 and 10 pphm are very common (Haering 1977, Indiana State Board of Health, unpub. data).

During June, July, and August of 1977, hourly maxima equal to or greater than 4 pphm were recorded 93% of the time at O_3 monitoring sites in Ashland, Kentucky; New Albany, Indiana; Springfield, Illinois; Youngstown, Ohio, and Indianapolis. Hourly maxima greater than 10 pphm were recorded at these sites 16% of the time (Teknekron, Inc., unpub. data), indicating that virtually all sensitive vegetation in the ORBES region is chronically exposed to potentially damaging concentrations of O_3 (see Figures V-4 and V-5 and Section VIII this report). Exposures for a duration of only eight hours at these concentrations has been shown to be sufficient to cause injury to eastern white pine.

Sulfur Dioxide

The combustion of coal is the major source of SO_2 releases to the atmosphere, accounting for approximately 60% of the total in the U.S. (U.S. Forest Service 1973). Most of the coal is burned to produce electricity, and in the ORBES region much of the coal is high in sulfur. Other sources of SO_2 are the refining and combustion of petroleum and natural gas, the smelting and refining of ores, and the manufacture of sulfur and sulfuric acid.

SO_2 effects on eastern white pines may be acute or chronic. After an acute exposure, the pines develop an orange-red (Applied Science Associates 1978, Linzon 1969), reddish-brown (U.S. Forest Service 1973), or orange-brown (Costonis 1970) necrosis of the current year's needles. Depending on the severity of the fumigation this necrotic discoloration may progress basipetally until the entire needle is dead. Often there is a sharp demarcation between the injured tips and the normally green bases (Linzon 1969). Gerhold (1977) notes that the accumulation of sulfur after fumigation with SO_2 is usually greatest in the tips of the needles, which may explain the necrosis of this part of the needle tips. With age, leaf tissue becomes more sensitive to SO_2. If successive fumigations occur, chlorotic bands may become visible (U.S. Forest Service 1973).

Chronic exposure to SO_2 is usually manifested as a general yellowing or chlorosis of the needles, along with premature defoliation (U.S. Forest

Service 1973, Applied Science Associates 1978, Linzon 1969). On older needles the chlorosis changes to a reddish-brown color, beginning at the tips of needles and again developing basipetally. Chronic injury will sometimes lead to tip necrosis.

Pines exposed to relatively high concentrations of SO_2 (e.g., near a point source) have fewer and lighter seeds than trees found farther from a source (Houston and Dochinger 1977). In addition, photosynthesis and net assimilation are depressed, and respiration and transpiration increased (Gerhold 1977). Numerous investigators (U.S. Forest Service 1973, Applied Science Associates 1978, Costonis 1970, Gerhold 1977) have noted that white pine needles are most sensitive to SO_2 (and other air pollutants) during a six-to-eight-week period when new needles are elongating (late spring/early summer).

Dochinger and Seliskar (1970) showed that chlorotic dwarf disease is caused by the interaction of SO_2 and O_3 on the foliage of genetically susceptible white pines. The symptoms range from a mild yellow mottling of the needles to severe stunting of all plant parts, with trees retaining only the current year's needles. The symptoms of chlorotic mottle are reversible when white pines are protected from SO_2 and O_3. In the Dochinger and Seliskar tests, sensitive trees receiving filtered air retained three years' growth of unblemished needles, whereas trees receiving 0.1 parts per million (ppm) SO_2 or 0.1 ppm O_3 showed flecking, slight to moderate mottling, and occasional shedding of older needles. When SO_2 and O_3 were combined, the effects were much greater than when they were administered alone, causing severe mottling of needles and premature needle abscission.

Synergistic interactions between SO_2 and O_3 that produce symptoms on white pine foliage have also been shown by Banfield (1972). Typical field symptoms include spotting and tip necrosis. In repeated 12-hour daylight exposure periods, Banfield found that O_3 at 0.1 ppm plus SO_2 at 0.05 ppm for 1-12 days induced blight on every tree exposed. Such concentrations are not uncommon in the ORBES region but they generally do not last for 12 hours (see Figure V-4 and Table IV-7).

Minor Pollutants and Agents that Cause Similar Symptoms

Some air pollutants cause symptoms similar to those caused by SO_2 and O_3 (U.S. Forest Service 1973, Applied Science Associates 1978, Skelly *et al*. 1972, Stone and Skelly 1974, Treshow 1969). These include fluoride, ethylene, ammonia, chlorine, hydrogen chloride, and oxides of nitrogen. The regional effects of these pollutants appear to be minor, although, locally, they can produce severe injury. A laboratory analysis of foliage constituents is helpful in the determination of causative agent(s) of injury.

Agents other than air pollutants also can produce symptoms on white pine similar to those caused by air pollution. Deficiencies of nutrients (i.e., nitrogen, iron, sulfur, copper, and boron) can lead to stunted growth, smaller leaves, general chlorosis of leaves, tipburn, and premature leaf drop (Applied Science Associates 1978, Cole et al. 1969). Other factors that may cause similar symptoms include salinity, soil acidity, pesticides and toxic chemicals, drought and/or flooding, low or high temperatures, insects, viruses and mycoplasmas, bacteria, and nematodes (Applied Science Associates 1978, Hepting and Berry 1961, Johnson and Lyon 1976). Williams (1975) states that agents such as bark beetles, needle cast fungi (Elytroderma deformans), dwarf mistletoe, natural senescence, Fomes annosus, scale insects, aphids, hail, drought, frost, nutritional disorders, gophers, limb rust, and weather fleck can cause difficulties in diagnosis of the causative agent of injury. However, he notes that O_3 mottle can be differentiated in the field from the effects of these other agents.

METHODS

Eastern white pines at nine stands in central and southern Indiana (Figure IV-1, Table IV-1) and four sites in Wisconsin (Figure IV-2) were examined for symptoms of air pollution stress, including chlorotic mottling/flecking, tip necrosis and chlorosis, and premature defoliation. These symptoms are commonly associated with O_3, SO_2 and/or hydrofluoric acid injury. Symptoms of winter injury, salts, hail, drought, scale insects, aphids, sawflies, case bearers, sooty molds, physical abrasion, herbicides, mites and spittlebugs were separable from the symptoms attributable to air pollution. No trees were impacted by winter de-icing salts.

Entire trees were examined for their general state of health. Symptoms of all types were assessed on 1-year-old needles from approximately ten different branches of all sides of each tree at a height of 1 to 3 meters. Needle retention for 2-year-old needles was determined for a height of at least 7 to 8 meters, and was estimated from branches in the upper and lower crowns. If uncertainty existed for symptom classification of the majority of needles, trees were placed in a less severe category. When doubt existed as to whether the symptoms could be attributed to air pollution, the pollution was not implicated in the assessment, although the classical symptoms of O_3 damage (chlorotic mottling and flecking) were nearly universal in the field, and varied only in intensity of expression. The results from one site (Bloomington 1979) were not included in any analyses because of suspected observer bias.

Pines at the nine stands in Indiana were examined by the senior author between 15 May to 5 June 1979, and again from 19 May to 30 May 1980. In addition, four stands in central Wisconsin (Figure IV-2) were examined for comparison, on 4 and 5 June 1980. These four stands were four of fifteen previously sampled by Karnosky (1980) from 1971-77.

Index of Injury

An index of injury was developed to summarize the state of health of individual trees and plantations (Table IV-2). This index was calculated by assigning numerical values to the continuous variables of air pollu-

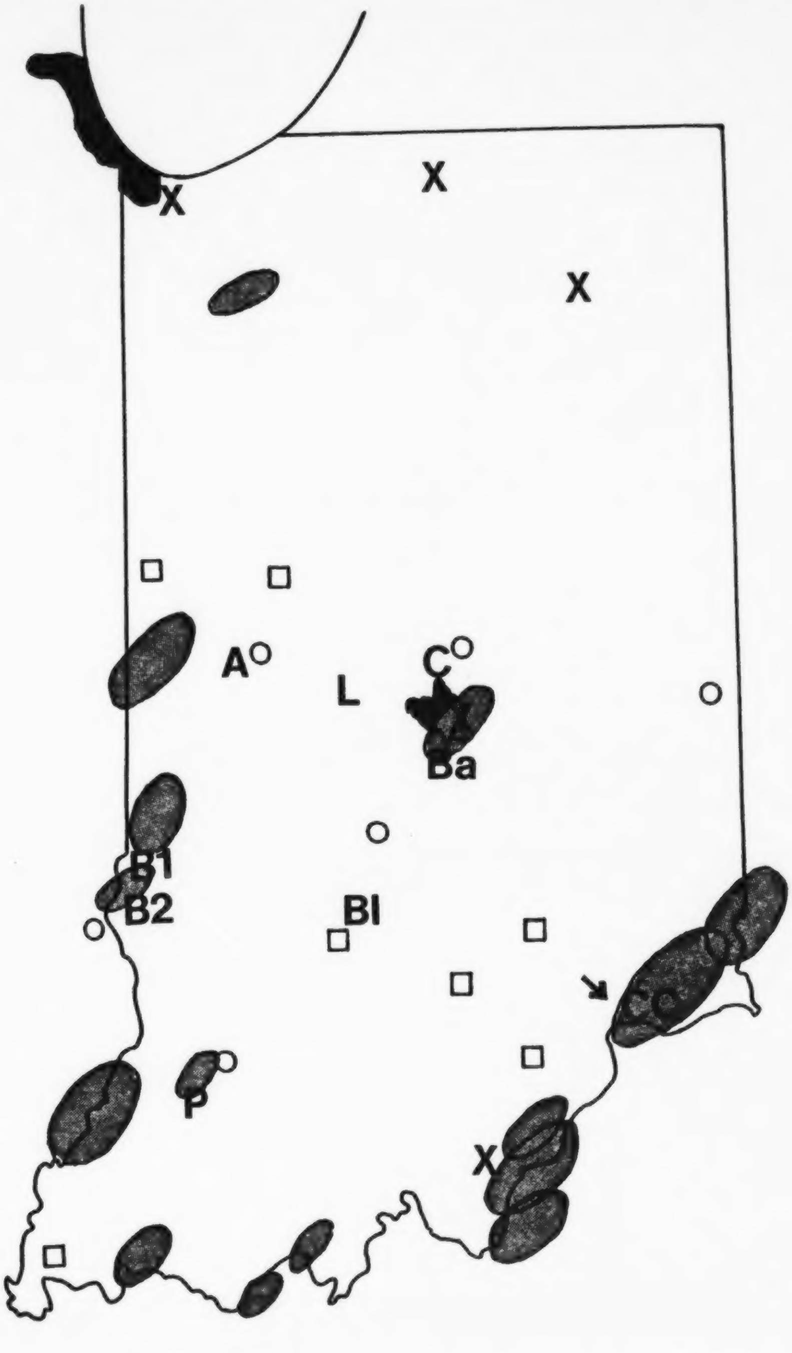

Figure IV-1. Location of nine eastern white pine stands surveyed in 1979 and 1980, in relation to point sources of sulfur dioxide. Shaded ellipses represent areas where SO_2 concentrations are calculated to reach 0.05 ppm twice per month around coal-fired power plant. Blank circles are fossil steam plants and blank squares are industrial sources of SO_2. Asterisks mark location of ozone monitors (see Table IV-6). Names of sites are given in Table IV-1.

TABLE IV-1. LOCATION OF WHITE PINES SAMPLED

Site Name	Location of Site	County
Bloomington (B1)	S29; T 8N; R 1W	Monroe
Alamo (A)	NE ¼, S26; T 18N; R 6W	Montgomery
Lizton (L)	W½, S33; T 17N; R 1W	Hendricks
Petersburg (P)	NW¼, NE¼, S22; T 1N; R 8W	Pike
Breed Site #1 (B1)	SE¼, S25; T 10N; R 10W	Vigo
Breed Site #2 (B2)	NW¼, SW¼, S5; T 5N; R 10W	Sullivan
Clifty Creek (CC)	NE¼ S29; T 4N; R 10E	Jefferson
Banta Road (Ba)	S10; T 14N; R 3E	Marion
Carmel (C)	S30; T 18N; R 4E	Hamilton

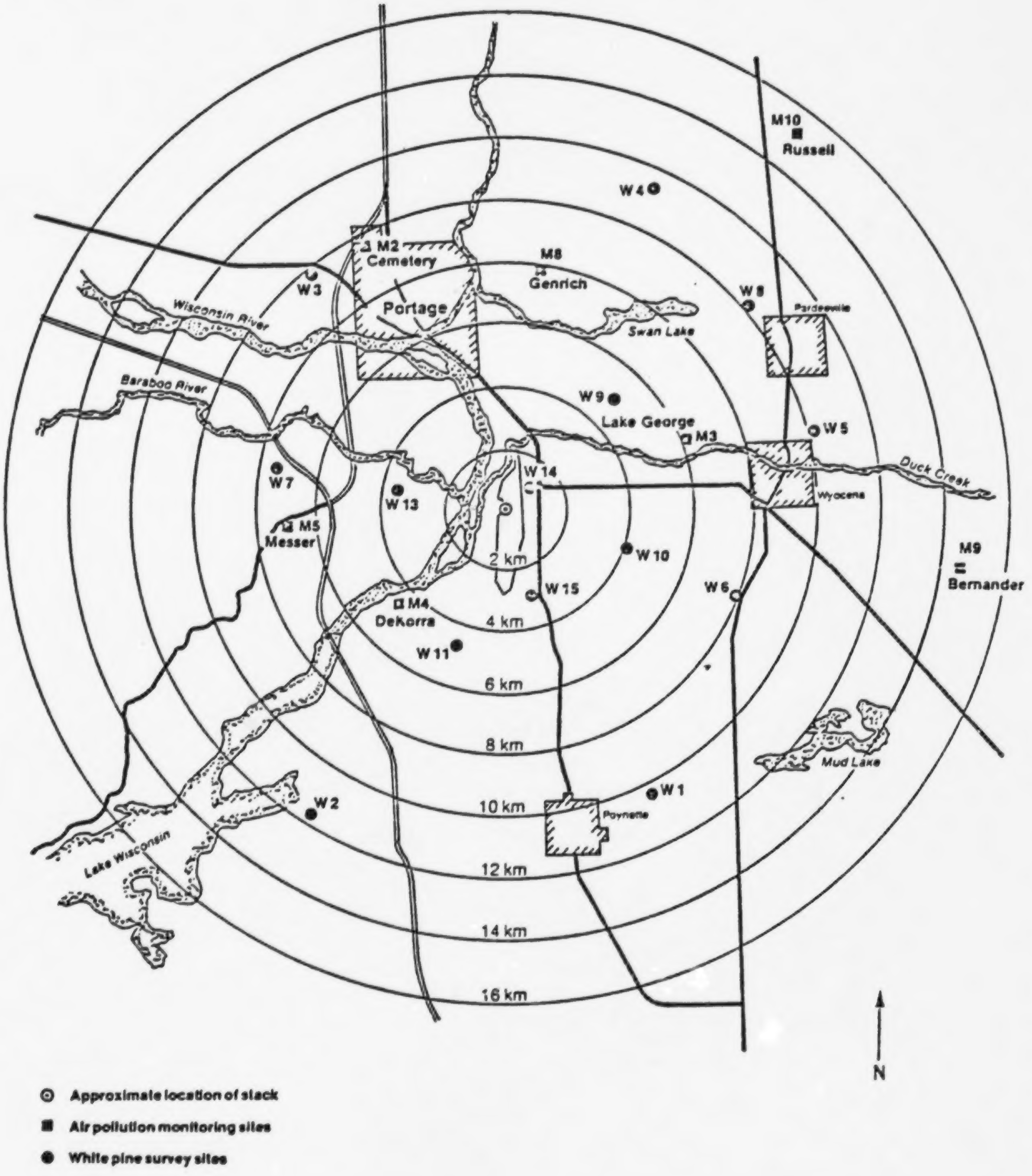
M10
Russell
W4
M2
Cemetery
Portage
M8
Genrich
W3
W8
Pardeeville
Wisconsin River
Swan Lake
Baraboo River
W9
Lake George
M3
W5
Duck Creek
Wyocena
W7
W 14
W 13
M5
Messer
2 km
W 10
M9
Bernander
W 15
W6
M4
DeKorra
4 km
W 11
6 km
8 km
Mud Lake
W 1
W 2
10 km
Poynette
Lake Wisconsin
12 km
14 km
16 km
N
Approximate location of stack
Air pollution monitoring sites
White pine survey sites

Figure IV-2. Location of eastern white pine stands around the Columbia Generating Station in Wisconsin, from Tibbitts et al. (1977). Stands numbered 3, 5, 9 and 14 were sampled in 1980.

TABLE IV-2. DISEASE INDEX FOR AIR POLLUTION SYMPTOMS ON EASTERN WHITE PINE

Numerical Value	20	40	60	80	100
Mottling/Flecking (majority of needles)	None	Very Slight	Slight	Moderate	Severe
Percent of Needles Necrotic	None	1-19%	20-49%	50-89%	90-100%
Length of Necrosis (majority of needles)	None	<1 mm	1 mm	1-5 mm	>5 mm
Retention	>60%	40-59%	20-39%	1-19%	None
Index Value	80	160	240	320	400
Disease Category	None	Slight	Slight	Moderate	Severe

tion-induced disease etiology. Mottling/flecking, for example, was assigned values of 20 (no symptoms on the majority of needles); 40 (very slight); 60 (slight disease); 80 (moderate disease), and 100 (severe disease). These values were then multiplied by the percentage of trees in a stand that belonged to that subcategory. Scores for each subcategory were summed to produce an index value for each category for each stand. The scores for each category then were summed to produce a total index of injury value for each stand. The indices for all types of air pollution-induced symptoms were given equal weight, with "80" being the lowest possible index, for "no visible air pollution disease", and 400 being the highest, for "severe disease." Individual trees also were scored for index of injury (Figure IV-3).

Statistical Tests

Due to the geographical extent of sampling sites and the known genetic variability of P. strobus with regard to air pollution sensitivity, disease index variances for the 13 sites in 1980 were tested for homogeneity via Bartlett's technique (Sokal and Rohlf 1969). The variances were shown to be homogeneous, demonstrating homoscedasticity, [x^2 of 10.45; critical x^2 .01(12) = 26.22]. In an initial analysis of variance, disease indices for each Indiana site were log-transformed. The resultant F-ratio was only slightly lower than non-transformed data and was still significantly different. Thus, the test populations are considered to be normally distributed and the original index values were analyzed.

Frequency distributions for disease indices (Figure IV-3) and each of the four principle components of the index (percent needle retention, percent needle necrosis, length of necrosis and degree of needle chlorosis/flecking) were graphed. The four components were then described for population skewness (coefficient of skewness-g_1) and g_1 statistics were tested for association with the overall index value via linear regression tests (Figures IV-4 to 7). In Figures IV-4 to 6, positive coefficients of skewness indicate fewer and less severe disease symptoms per sample and negative coefficients (g_1) indicate a more severe disease situation per component measured in the site population. A zero g_1 indicates a normal distribution which in the case of flecking/chlorosis means that most trees were categorized as having slight mottling and flecking in the majority of needles (Table IV-2 and Figure IV-4 "Carmel").

These techniques allow the distribution of tree health within each site, for each principal component, to be assessed, and if a gradient of disease severity exists between sites, quantification of the differences can be described and evaluated.

Pollutant Concentrations

O_3 concentrations during June to August 1977 for stations in Illinois, Indiana, Kentucky, and Ohio were obtained from Teknekron Research, Inc. Ozone data for five sites in Indiana for May to August of 1978 and 1979 were obtained from the Air Pollution Division of the Indiana State Board of Health. Ozone data for Portage, Wisconsin, from June to August

1976-1979, were obtained from the Water Resources Center of the University of Wisconsin at Madison. Concentrations of SO_2 for June to August 1975-77 in Indiana were obtained from Teknekron Research, Inc., and from the Air Pollution Division of the Indiana State Board of Health for 1977. The SO_2 concentrations around the Columbia Generating Station in Portage, Wisconsin, from June-August 1977 were obtained from the Water Resources Center of the University of Wisconsin-Madison.

SITE DESCRIPTIONS

The locations of the nine Indiana stands of eastern white pine are shown in Figure IV-1 and Table IV-1, in relation to major utility and industrial sources of SO_2.

Bloomington Site

This site is located 2 miles past the southern edge of Bloomington in a residential area of slightly rolling hills. The trees are exposed primarily from the north, with limited southern exposure.

Alamo Site

Located 1 mile south of the town of Alamo and 400 feet west of County Road 825W, this site is bound on three sides by a cornfield and on the west side by a county road. The plantation is exposed from all directions.

Two coal-fired power plants are in the vicinity of this site. A 25-megawatt (MW) plant is located 10 miles northeast of the stand, and a 1,062 MW plant is 22 miles due west of the stand. Twenty trees on the east and twenty on the west side of the stand were sampled.

Breed #1

This site is located on State Road 246, approximately 1.6 miles east of its intersection with State Road 63 in Prairie Creek. The site is about 8 miles northeast of a 500-MW coal-fired power plant with prevailing winds during the summer from the southwest. When the wind is from the north, this stand is in line of the Terre Haute urban plume. However, the impact of the Terre Haute plume (and industrial sources located to the north) is considered insignificant, since the pines are protected on the north by a stand of deciduous trees that probably scavenge the pollutants. The topography surrounding this stand is flat, with the majority of land used for farming. A small pond is in the middle of the stand.

Petersburg

This site is located on the northern edge of Petersburg, 1 to 1.5 miles west-southwest of two coal-fired power plants with a combined capacity of 1456 MW. A 1300-MW coal-fired power plant is approximately 25 miles west-southwest, and a 230-MW coal-fired plant is about 10 miles to the south.

The white pines sampled are in a residential area of gently rolling hills. The trees are not in one definitive stand, as at other locations, and are not of the same age. The trees are in groups of two or three in yards of different houses. About half of the sampled trees are completely exposed to ambient air conditions, and the other half are protected to varying degrees by surrounding vegetation.

Clifty Creek

This stand of trees is located within Clifty Falls State Park. The large majority of trees are red pine (*Pinus resinosa* Ait.), with eastern white and jack pine (*Pinus banksiana* Lamb), also present. Thirty-four red pines and 20 white pines were sampled.

The stand is surrounded on all sides by deciduous forest on fairly steep terrain. The stand has a sloping southern half and a flat northern half. Overall, this stand of trees is more protected from air pollutants by surrounding vegetation than other stands.

The trees are located approximately 2 miles north of the Clifty Creek power plant, which has a 1300-MW capacity and an average yearly SO_2 emission rate of 8,620 grams/second (gs^{-1}) (Teknekron, Inc. 1979).

Breed #2

This stand is located 1.5 to 2 miles north-northeast of the Breed power plant, and is within the calculated 0.05 ppm SO_2 impact zone associated with this plant (see Section VI). As noted in the site description of Breed #1, several industrial sources in the area contribute to the level of air pollution.

The stand is exposed from all directions except the north, where a stand of deciduous trees probably protects the pines from pollutants originating from the north (i.e., from Terre Haute and industrial sources). The surrounding topography is flat, agricultural land. The trees grow in a north-south line.

Lizton

This stand is 1 mile south of Lizton on State Road 39, 300 feet east of the road. The stand consists of four rows of white pine growing in a north-south direction. The stand is surrounded by predominantly flat pastureland, with good exposure from all sides. The closest known air pollutant sources are Indianapolis, approximately 20 miles east-southeast of the stand, and an oil refinery 15 miles due east.

Banta Road

This site is located on the southwestern edge of Indianapolis, 2 to 2.5 miles due south of a 900-MW coal-fired power plant. The trees are also exposed to the general Indianapolis plume of pollutants.

There are two north-south rows of pines in this stand; the easternmost row was sampled, so the trees sampled are directly exposed to pollutants, primarily from the east. Five or six branches on the exposed east half of each tree were sampled.

Carmel

Carmel is a northern suburb of Indianapolis, and the site is located along the east and south sides of the Carmel Public Library. The topography of the surrounding residential area is flat, and the site is open on all four sides.

Wisconsin Sites

The sites sampled around the Columbia Generating Station near Portage, Wisconsin, correspond to the stands sampled by Tibbitts *et al.* (1977) (Figure IV-2). Stands #3, 5, 9 and 14 were sampled in 1980. All trees sampled had good exposure to ambient conditions.

RESULTS

Table IV-3 summarizes the indices of foliar injury derived from the data. The mean index of injury value is 297.5 for the eight stands in Indiana for 1979 and 268.5 for the nine stands in 1980. This corresponds to moderate disease. The mean index value is 162.7 for the four Wisconsin sites for 1980, corresponding to slight disease. Indices for individual stands in Indiana and Wisconsin over the 2-year period range from 132.7 (rural Wisconsin) to 333.6 (urban Indiana). Individual tree ratings range from 80 to 400.

Symptoms of air pollution disease were present at all locations sampled (Tables IV-4 and 5). At the Indiana sites, 99.6% of the pines displayed symptoms of air pollution disease, while in Wisconsin 92.5% displayed symptoms. There were significant differences in disease intensity between sites [$F = 22.43$; $F_{.001}$ $(12,360) = 2.74$] and, for the Indiana sites, air pollution disease was significantly worse in 1979 than in 1980 [$F = 5.93$; $F_{.05}$ $(1.16) = 4.49$; $F_{.025}$ $(1.16) = 6.12$] (Table IV-5). The Duncan's Multiple Range Test was more sensitive than the Student-Newman-Keuls (SNK) test to discern *a posteriori* differences between site indices. According to the SNK method, no site means were different from year to year, but the Duncan's test showed that the Breed #2 and Lizton sites were significantly more diseased in 1979 than in 1980. These differences corresponded to different air pollution loads (Table IV-6) during the growing seasons of the two years.

Trees were significantly healthier in central Wisconsin than in Indiana (Figure IV-3, Tables IV-4 and 5) when site indices were compared with Duncan's and the SNK multiple range tests. Duncan's Multiple Range Test appeared more sensitive than the SNK tests to separate disease intensity differences at different sites (Table IV-5). The sites represented a gradient of disease intensity, with two types of disease intensity suspected in Wisconsin and two types suspected in Indiana (Table IV-5).

In Indiana over the 2-year period, 80.5% of the trees showed tip necrosis on 20% or more of their needles, whereas only 13.8% of the pines observed had no needle necrosis (Table IV-4). Fifty-four percent of the trees had greater than 1 mm of tip necrosis on the majority of their needles, and 81% of the trees had symptoms of chlorotic mottling or fleck-

TABLE IV-3. INDEX OF FOLIAR INJURY FOR 1979 and 1980 CALCULATED FROM MOTTLING/FLECKING, NECROSIS, AND NEEDLE RETENTION CATEGORIES RATED ON A 20 TO 100 POINT SCALE. INDEX RANGE IS 80 TO 400, WITH 80 = NO VISIBLE DISEASE. AND 400 = SEVERE DISEASE.

			1979			
Site	# Trees Sampled	Mottling/ Flecking	Percent Needles Necrotic	Length Tip Necrosis	Needle Retention	Total
Alamo	40	34.0	76.4	61.5	85.0	256.9
Clifty Creek	20	69.0	59.0	55.0	90.0	273.0
Breed #1	26	56.2	76.0	43.8	98.4	274.4
Petersburg	21	63.8	80.0	47.5	89.5	280.8
Banta Road	24	70.8	81.4	70.0	97.6	319.8
Lizton	32	64.9	92.4	69.9	93.8	321.0
Breed #2	16	77.6	87.4	63.7	92.6	321.3
Carmel	25	66.4	96.0	80.8	90.4	333.6
			1980			
Wisconsin 3	30	28.8	23.9	21.4	58.6	132.7
Wisconsin 14	30	27.4	30.0	22.8	63.5	143.7
Wisconsin 9	30	22.7	48.7	35.3	67.3	174.0
Wisconsin 5	30	30.1	51.4	38.6	80.6	200.7
Bloomington	49	44.9	63.6	53.3	82.0	243.7
Alamo	40	52.0	65.0	50.0	77.0	244.0
Clifty Creek	19	62.1	51.7	43.2	87.2	244.2
Breed #2	16	79.9	66.2	42.6	81.3	270.0
Breed #1	26	59.3	63.8	57.7	89.9	270.7
Lizton	32	46.8	76.6	61.7	87.4	272.5
Petersburg	22	56.1	78.8	63.4	79.9	278.2
Banta Road	24	41.7	86.6	65.8	91.6	285.7
Carmel	25	48.8	95.2	84.0	79.2	307.2

TABLE IV-4. PERCENT OF EASTERN WHITE PINE AT 9 STANDS IN INDIANA AND 4 IN WISCONSIN WITH VARIOUS SYMPTOMS ATTRIBUTABLE TO AIR POLLUTION

Symptom	(1979-1980) Indiana, n = 457	(1980) Wisconsin, n = 120
Chlorotic Mottle/Fleck on majority of needles:	%	%
None	18.8	74.2
Very Slight	22.1	18.3
Slight	30.0	5.0
Moderate	19.9	2.5
Severe	9.2	0.0
Tip Necrosis on:		
None of foliage	13.8	67.5
Over 20% of foliage	80.5	26.7
Majority of foliage	70.6	21
Length of tip necrosis based on majority of needles:		
None	29.4	79
1 mm or less	16.7	15.8
1 - 5 mm	45.8	5.0
Over 5 mm	8.1	0.0
Percent of second year needles retained:		
No retention	55.8	18.3
1-19%	30.8	30.8
20-39%	8.7	30.0
40-59%	3.0	11.7
>60%	1.5	9.1

TABLE IV-5. MULTIPLE RANGE ANALYSIS OF SITE MEANS IN 1980 USING DUNCAN'S (DUN) AND STUDENT-NEWMAN-KEUL (SNK) METHODS

Geographic Aspect	Site	Mean Disease Index per Site 1980	1979	Disease Ranking
		DUN[1] SNK[1]		
Rural	Wisconsin 3	132.7		
Rural/Power Plant	Wisconsin 14	143.7		Slight
Rural/Power Plant	Wisconsin 9	174.0		
Rural/Power Plant	Wisconsin 5	200.7		
Rural	Bloomington	243.7	----	
Rural	Alamo	244.0	256.9	
Rural/Power Plant	Clifty Creek	244.2	273.0	Moderate
Rural/Power Plant	Breed 2	270.0	321.3[2]	
Rural/Power Plant	Breed 1	270.7	274.4	
Rural	Lizton	272.5	321.0[2]	
Rural/Power Plant	Petersburg	278.2	280.8	
Urban/Power Plant	Banta	285.7	319.8	
Urban	Carmel	307.2	333.6	Severe

[1] Brackets to the left of the column represent maximum non-significant ranges (p=0.05); brackets to the right of the column represent non-significant studentized ranges.

[2] Significant differences between years according to Duncan's Multiple Range Test, but not according to the Student-Newman-Keul MRT.

TABLE IV-6. NUMBER OF DAYS WITH OZONE EPISODES[1] AT OR GREATER THAN 0.08 PPM AT FIVE SITES IN INDIANA AND TWO SITES IN WISCONSIN, AND NUMBER OF HOURS OVER 0.08 AND 0.10 PPM DURING MAY-AUGUST 1978-1979

Sites	Number of days with episodes ≥ .08 ppm	hr. ≥ .08 ppm ozone	hr. ≥ .10 ppm ozone	Percent of days with data
Indianapolis Womens' Prison				
1978	38	301	63	100
1979	16	117	8	100
New Albany				
1978	19	148	38	72
1979	4	28	1	61
Hammond				
1978	38	259	67	97
1979	28	75	8	87
South Bend				
1978	8	65	9	52
1979	13	116	26	81
Fort Wayne				
1978	27	182	33	48
1979	0	7	5	57
Wisconsin				
1979 (Genrich)	16	114	16	100
1979 (Messer)	33	318	95	100

---

[1] Episode = 3 or more continuous hours at or above .08 ppm.

ing on a majority of needles (Table IV-4). Most trees with needle necrosis had from 1 to 5 mm dead tissue. In both years, 8% of the trees at all sites had needle necrosis greater than 5 mm and were classified as "severely" diseased (Table IV-2). On sensitive trees in Indianapolis, one-year-old needles were observed to become diseased and abscised as early as 22 June 1980. Up to that time the mean maximum hourly O_3 concentrations did not exceed 206 micrograms/cubic meter ($\mu g/m^3$) (0.105 ppm) and for SO_2 did not exceed 66 $\mu g/m^3$ (0.024) (Indianapolis Air Pollution Board 1980).

The red pines sampled at Clifty Falls State Park were uniform in their response to the 1978 and 1979 pollutant loads. Overall, the severity of air pollution-induced disease increased with increasing age of foliage. As a general rule, the 1-year-old foliage showed none to very slight injury, the 2-year-old foliage very slight to slight injury, and the 3-year-old foliage slight to moderate injury. In both years the trees retained only three years foliage, instead of a normal 4 to 5 year retention (Sargent 1965). The 1978 foliage, first surveyed in 1979, generally accumulated more severe injury in the mottling/flecking and tip chlorosis categories in 1980.

Frequency Distribution Analysis

At the Wisconsin sites, the majority of trees had only slight symptoms of air pollution disease. More severely diseased was a subpopulation of 14% of trees (Figure IV-3). These more severely diseased trees appear randomly in the stands and apparently constitute hypersensitive genotypes as encountered in other studies (Gerhold 1977, Karnosky 1980).

In Indiana, the frequency distribution of the disease index values more clearly approached a normal distribution in 1980, but were skewed toward more disease in 1979 (Figure IV-3). A remnant subpopulation of 17% remained resistant in 1979, but fluctuated in disease expression in 1980. The fluctuation in disease expression would be expected if the pollution load changed or if environmental conditions were more conducive to disease expression in 1979.

Flecking/Chlorotic Mottling

The symptoms of flecking and chlorotic mottling observed on one-year-old foliage of *P. strobus* closely approximate the classic initial manifestation of oxidant air pollution-induced disease on many kinds of conifers (Williams 1980, Phillips *et al*. 1977, Dochinger and Seliskar 1970, Anderson and Dochinger 1978). At all sites, one-year-old needles (second complement whorl) were a better indicator of oxidant injury than current year (the first and youngest rank of foliage). To illustrate this point, in Indianapolis on 27 June 1980, an air pollution episode occurred with 0.135 ppm O_3 and 0.01 ppm SO_2. The second complement of foliage (one year old) became typically chlorotic-mottled, and rapidly yellowed within a few days. The current-year needles were just completing elongation and had only minor chlorotic mottling, tip chlorosis, and tip burning. During the next thunderstorm, winds blew off most of the yellowed needles of the

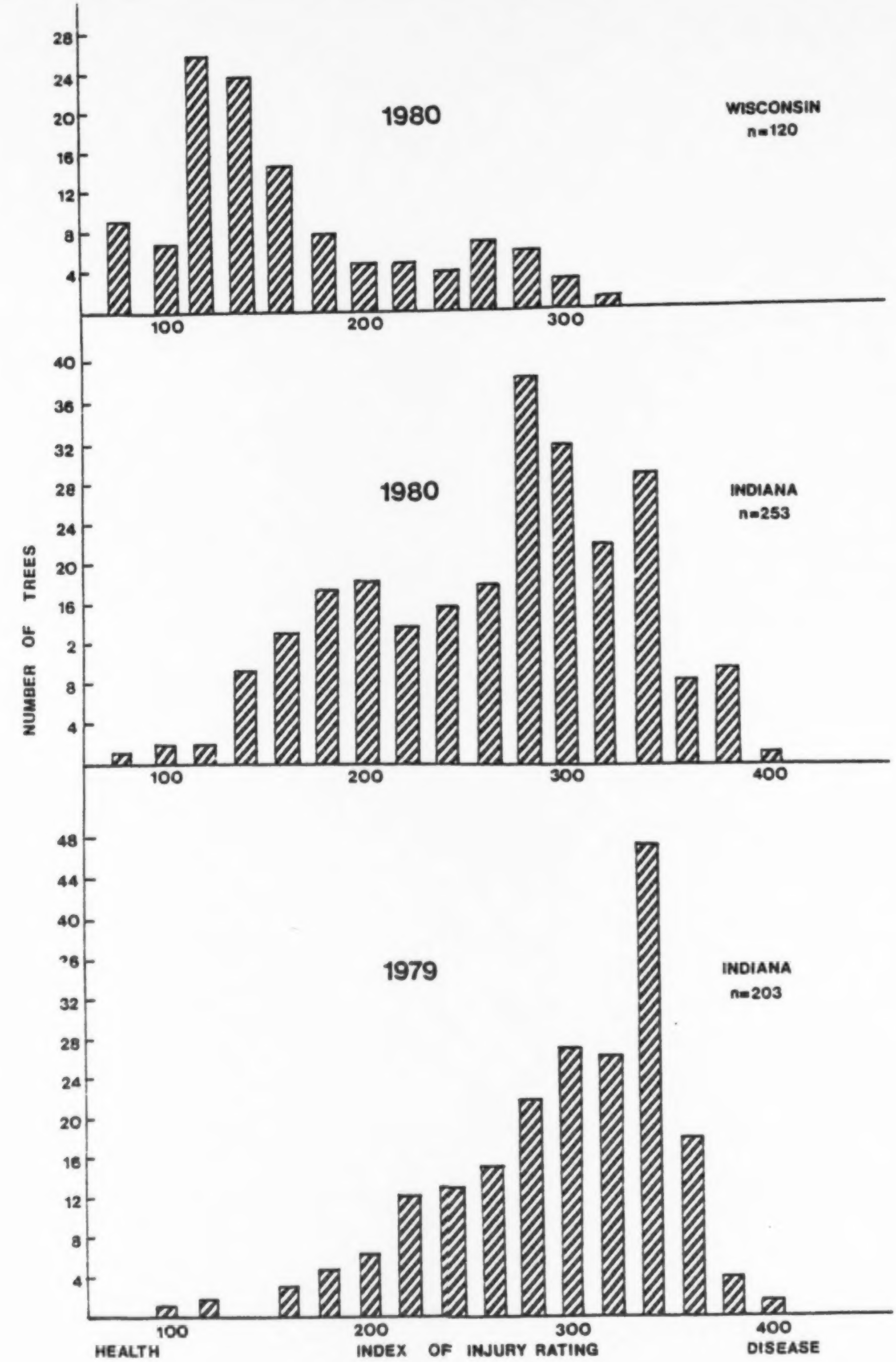

Figure IV-3. Frequency histogram of disease index scores for Indiana sites during 1979 and 1980, and four Wisconsin sites in 1980.

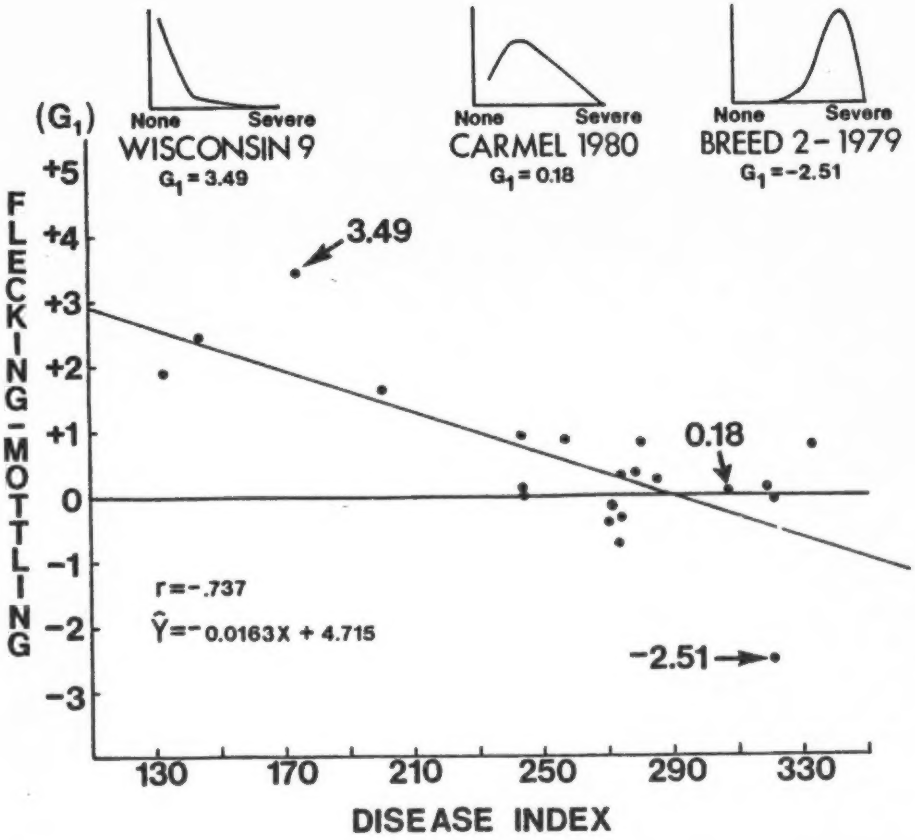

Figure IV-4. Linear regression of the coefficients of skewness (g_1) for the symptom of flecking/chlorotic mottling and disease indices for 13 sites in Indiana (1979-1980) and Wisconsin (1980). Inserted histograms are for sites in Wisconsin, Carmel and Breed. Their g_1 statistics are indicated by arrows.

second complement, whereupon the trees appeared to become greener due to the abscission of the more diseased tissue. This is a suggested mode of action for premature leaf fall in white pine and other conifers (Williams et al. 1977, Williams 1980).

The amount of flecking/chlorotic mottling within sites increased with increasing disease severity (Figure IV-4). More trees per site had severe flecking and chlorotic mottling in those sites with a greater disease index.

Percent Necrosis of Needles

A strong linear gradient exists in percent necrosis of needles of trees attributed to air pollution ($r = 0.899$) between the Wisconsin sites (healthy) and the Indiana urban sites (diseased). The two healthiest Wisconsin sites had the least needle necrosis (Figure IV-5). The more urban sites in Indiana, with more complex and concentrated air pollution, tended to have more needle necrosis. Frequency distribution within stands strongly tended to shift towards more severe disease as the disease index, and presumably the pollution load, increased.

The mean g_1 for the four Wisconsin sites is 1.67, and 67.5% of the one-year old needles had no needle necrosis. Among the Wisconsin trees, 12.5% had 90-100% of their needles with some degree of necrosis, and the resulting histogram for distribution of necrosis was strongly skewed (Figure IV-5).

The Indiana sites had a g_1 value of -0.932, indicating that the majority of the white pines had necrosis. The Lizton 1980 site (g_1 of -0.9233) is graphed for comparative purposes in Figure IV-5. At Lizton, 32.5% of the trees had necrosis to some degree on 90-100% of their needles, but only 9.3% had no necrosis. Tip necrosis is considered to be an initial symptom of air pollution disease on eastern white pine (personal communication, J. Skelly and D. Karnosky). This is confirmed, in part, by the fact that fewer trees have chlorotic/fleck necrosis than have tip necrosis (Table IV-4).

Length of Necrosis

The length of needle necrosis increased with the severity of disease, and strong shifts in frequency distribution occurred in the amount of needle necrosis length along a gradient of the Wisconsin rural and rural-to-urban sites in Indiana. The linear regression coefficient accounts for 80% of the variation about the line ($r = -0.80$).

The mean g_1 statistic for length of necrosis for the Indiana sites is -0.383. If the Breed #1 (1980) and Lizton (1980) coefficients are averaged (-0.376), the populations are seen to be divided into two reaction phenotypes; 32.6% of the population had no necrosis, and 65.5% of the population had needle necrosis greater than 1 mm (Figure IV-6 insert).

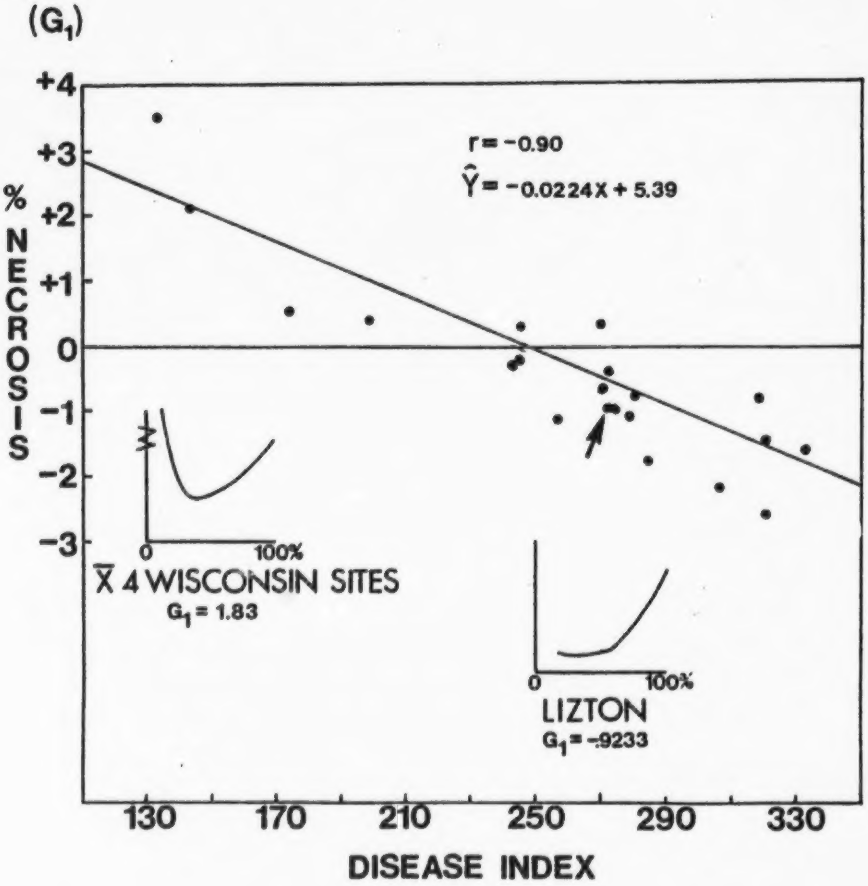

Figure IV-5. Linear regression of skewness coefficients (g_1) for percent of needle necrosis and disease indices for 9 sites in Indiana (1979-1980) and 4 sites in Wisconsin (1980). Inserted histograms are for the mean g_1 of the Wisconsin sites and at Lizton, Indiana. The Lizton g_1 is indicated by the arrow.

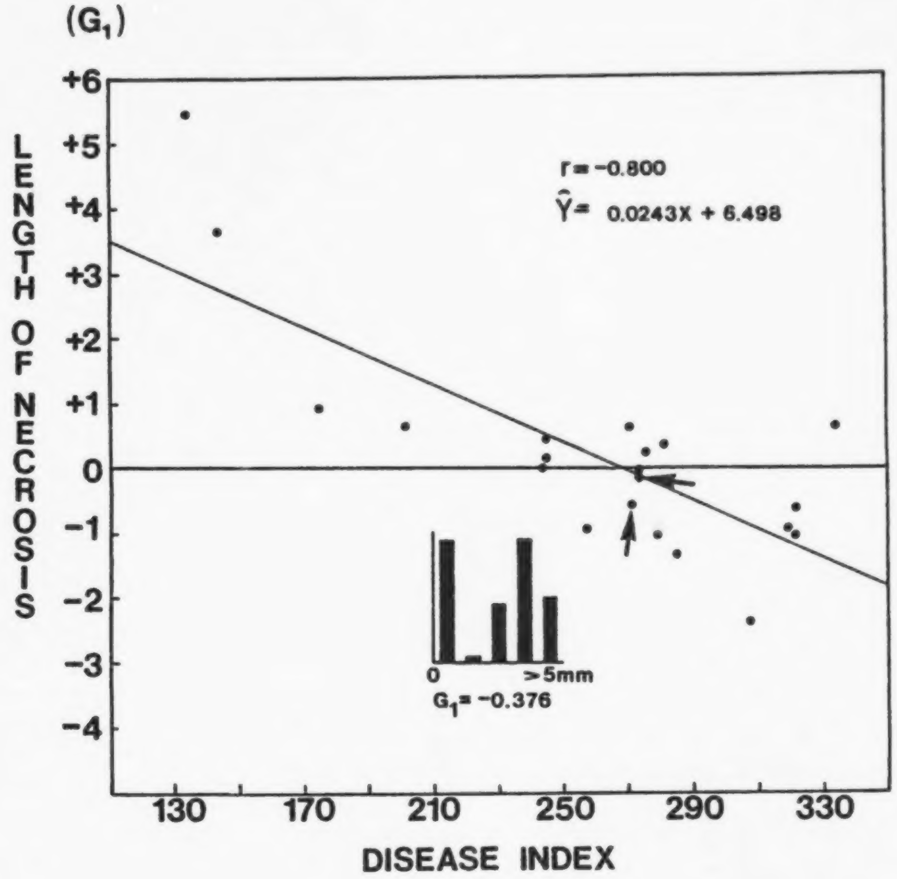

Figure IV-6. Linear regression of skewness coefficients (g_1) for length of needle necrosis and disease indices. Inserted histogram represents the mean g_1 statistic from the Breed and Lizton sites (arrows).

The Wisconsin sites had five times as many trees without tip necrosis as the Indiana sites (Table IV-4) and over ten times fewer trees with necrosis over 1 mm in length. On a comparative basis, the Wisconsin trees would be classified as slightly diseased and therefore less prone to long-term decline from additional air pollutants and predisposing factors than the more severely injured trees growing in Indiana.

Needle Retention

The relationship between increasing disease severity and increased premature needle abscission does not appear to be strong. At all sites, there is a tendency for premature leaf drop (which would be expected if O_3 were uniform over the region), with slightly more needle retention in Wisconsin. The correlation coefficient for the linear regression of needle retention X disease index is only 0.435, and although F-tests indicate a significant slope towards less retention in sites with more air pollution, premature needle abscission from air pollution appears to be a general phenomenon in the region. This is demonstrated in Figure IV-7, where all g_1 coefficients are positive, indicating a general skewness towards less than 20% retention of 2-year-old needles. Because the needles begin to fall early in the growing season after O_3 episodes, the presence or absence of needles is relatively more independent in predicting disease severity than the other variables which depend on the presence of needles.

For all sites, the mean g_1 is 1.058. One site near the overall mean is Petersburg (1.088), and that histogram is inserted in Figure IV-7. At the Petersburg site, 38% of the trees had no needle retention of 2-year-old foliage, and 33% of the trees had less than 19% needle retention of the 2-year-old needles. Gerhold (1977) reports that normal needle retention is 27 months for white pine. At the Indiana sites, 86% of the trees had less than 20% of their 2-year-old needles, whereas at the Wisconsin sites 49% of the trees had less than 20% retention of 2-year-old needles (Table IV-4). If needle retention is extremely sensitive to O_3-induced effects, it is possible that sites outside of the high O_3 region in the central U.S. may be required to evaluate its usefulness as an indicator.

Ambient Air Pollution Concentrations in Indiana

Ozone in Indiana, the Wisconsin sites, and the Ohio River Basin is commonly measured at concentrations and durations greater than that reported to cause needle chlorosis and necrosis on eastern white pine (Gerhold 1977) (Tables IV-6 and 7). Concentrations of O_3 in the range of 0.07 to 0.21 ppm occur with regularity during pollution episodes in rural areas of the Ohio River Basin (Bowen 1978, Lioy and Samson 1979, Haering 1977). During the growing season, peak O_3 values occur during periods of gradual buildup of O_3, generally after the slow passage of high-pressure systems. The aggregate of this process leads to development of a deep regional blanket of heavily polluted air which moves slowly across the United States east of the Mississippi River with regular frequency (24 times in 1976 and 16 episodes in 1977, according to Lioy and Samson 1979). Ozone was consistently of higher concentration and duration in 1978 than

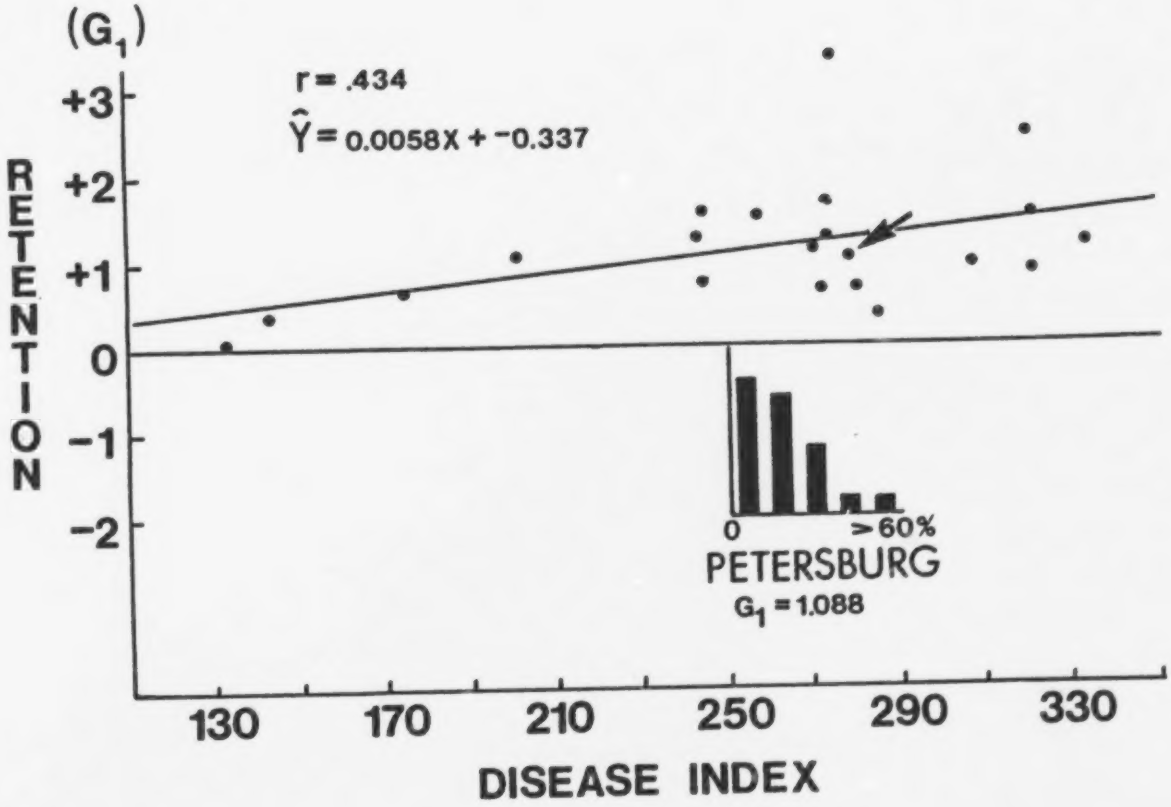

Figure IV-7. Linear regression of skewness coefficient (g_1) for needle retention and disease indices at sites in Wisconsin and Indiana. Inserted histogram represents the frequency distribution at the Petersburg, Indiana, site (g_1=1.088) and is indicated by the arrow.

TABLE IV-7. OZONE AND SULFUR DIOXIDE CONCENTRATIONS AT KEY MIDWESTERN U.S. LOCATIONS.

	OZONE				SULFUR DIOXIDE	
	Mean of Maximum Hourly Averages June-Aug. 1977	Number of Days Where Maximum Hourly Mean Equals or Exceeds				
Location	(ppm)[a]	.05	.10	.12 ppm	Location	Mean Concentration
Springfield, Ill.	0.0679	57	7	2	Clifty Creek	0.131 ppm (344.4 μg/m^3)[c]
New Albany, Ill.	0.0856	80	23	13	Breed #2	0.088 ppm (230.9 μg/m^3)[c]
Indianapolis, Ind.	0.0745	61	17	7	Breed #1	0.076 ppm (200.0 μg/m^3)[c]
Youngstown, Ohio	0.0730	76	13	4	Banta Road	0.071 ppm (186.5 μg/m^3)[d]
Ashland, Ky.	0.0592	--	--	--	Petersburg	0.088 ppm (230.9 μg/m^3)[e]
Portage, Wisc.[b]					Portage, Wisc.	0.029 ppm (75 μg/m^3)[f]
1976-Genrich	0.0607	--	--	--		
1977-Genrich	0.0547	--	--	--		
1978-Genrich	0.0659	--	--	--		

[a] From Teknekron Research, Inc., Berkeley, California.

[b] From Water Resources Center, University of Wisconsin-Madison.

[c] From Teknekron Research, Inc. Values represent the mean of the 9 second-highest hourly observations/month from June, July and August of 1975, 1976, and 1977.

[d] From Indiana State Board of Health, Air Pollution Division. Mean of 4 second-highest hourly observations per month from Jan., Apr., May and June 1977.

[e] Calculated to be equivalent to Breed #2 due to same location within calculated area of sulfur dioxide impact. See Sections VI and VII, this report.

[f] From Water Resources Center, University of Wisconsin-Madison. Value represents mean of the 3 second-highest hourly observations per month from June, July and August 1977 at seven monitoring stations.

in 1979 (Table IV-6), and the index of injury reflects this air pollution difference (Figure IV-3, Table IV-3).

Potentially phytotoxic SO_2 concentrations have been measured repeatedly in numerous locations in Indiana (Teknekron, Inc. 1979, Haering 1977). The urban areas around southern Indiana and Indianapolis have levels of SO_2 that regularly exceed 200 $\mu g/m^3$, and some are greater than 365 $\mu g/m^3$ (0.074-0.135 ppm) (U.S. HEW 1969, U.S. EPA 1977). In addition, zones of SO_2 concentrations downwind from 15 Indiana coal-fired power plants have been calculated using peak-load emission rates by Miller and Wong (Section VI). Four, and possibly five, of the nine study plantations lie within or adjacent to these zones (Figure IV-1), where SO_2 is estimated to reach 0.05 ppm for at least the second highest hourly observation per month from June through August. Mean estimated concentrations for SO_2 at several plantations are listed in Table IV-7.

The measured concentrations of air pollutants, alone or in combination with each other or with other phytotoxic pollutants such as NO_x are sufficient to elicit the observed disease reactions in white pine over large areas of Indiana.

In Wisconsin, O_3 levels at the two O_3 monitors closest to the study plots vary considerably, with the Messer station (at higher elevation) consistently higher than the Genrich station. The Genrich station O_3 dose for 1979 is similar to Indianapolis concentrations in 1979, but the Messer 1979 dose is substantially higher. These two stations are about equidistant from the study plots, with the Messer site situated on a ridge 150 meters higher in elevation than the four study sites. According to Karnosky (1980), O_3 and SO_2 concentrations in the range of 0.004 to 0.05 ppm for 1-3 hours during the summer months are common near the Wisconsin sites. The primary difference in air quality between the Indiana and Wisconsin sites appears due to the frequency and duration of slightly elevated SO_2 present.

DISCUSSION

Sufficiently high concentrations of air pollutants exist in rural and urban Indiana to cause the extensive foliar injury observed on eastern white pine. Since 99.6% of the trees examined developed symptoms of disease attributable to air pollution, tolerant as well as sensitive genotypes of white pines are being affected. Chlorosis, necrosis and premature needle abscission, which result from air pollution stress, cause ornamental and wind shelter values to be lowered and reduce wood fiber production in forests and plantations. Premature abscission of needles after only 14 - 17 months, as observed, instead of the normal 27 months (Gerhold 1977) could reduce wood accretion by 25%. The loss would be greater with consideration of pollution-induced metabolic stress on the first and second years of foliage, which would also slow or prevent wood accretion (Keller 1980). Such an effect seems probable, because fascicular needles continue to provide substantial amounts of assimilates in their second year, and may photosynthesize at up to 65% of their first-year rates into their third year, just before abscission (Zelawski 1976). Zelawski stresses that needle retention is very important for productivity. Kozlowski and Winger (1964) report that in P. resinosa 1-year-old needles were the major source of carbohydrate for shoot growth and accounted for four-fifths or more of all shoot growth. In addition, Wilkinson and Barnes (1973) noted altered CO_2 fixation patterns on white pine with O_3 doses as low as 0.10 ppm for 10 minutes. Ozone in Indianapolis had exceeded 0.10 ppm for 4 hours by 27 June 1980, the critical period for needle injury according to observations of Karnosky (1980), Dochinger and Seliskar (1970), Phillips et al. (1977), and Skelly et al. (1972).

Fluctuations in disease severity at some of the sites were not as pronounced as expected. Stands in rural Wisconsin were not symptom-free, and the gradient of disease severity in Indiana was very gradual. The more rural sites distant from known point sources were generally the most healthy, but by a slight margin. This may simply reflect the apparent dominance of O_3-induced symptoms, although local and remote sources of SO_2 can be seen as a factor. Urban plantings in polluted surroundings had the most severe disease condition.

The disease index is sensitive to differences in O_3 fluctuations from year to year. Oxidants in 1978 were worse in Indiana than 1979, and the air pollution disease on white pine also was significantly worse on 1978 than 1979 needles (surveyed in 1979 and 1980). Changes in apparent tree health also could be caused by strong winds and driving rains that dislodge air pollution-stressed fasicles, leaving only healthier foliage. Thus, the observed damage may represent only a portion of the total air pollution damage.

Despite similar amounts of O_3, there were suprising differences in air pollution disease expression between Wisconsin and Indiana. Four factors may be responsible: (1) The Indiana sites appear to have more SO_2 than the Wisconsin sites,; (2) Indiana is in the southern part of the range of white pine, so the trees in this state may be more apt to show visible disease expression, especially premature needle abscission; (3) urban areas have more NO_x, which tends to quench O_3, but can react in combination with SO_x on vegetation -- the five O_3 monitors in Indiana are in urban areas, whereas the Wisconsin monitors are in rural settings; and (4) greater relative humidity in Indiana than Wisconsin may predispose trees to air pollution injury.

The widespread occurrence of air pollution symptoms on eastern white pine is evidence that O_3 is affecting vegetation on a regional basis. This is supported by observations of visible air pollution symptoms in Indianapolis on red pine, Norway spruce, Colorado blue spruce, sycamore, Tulip poplar, sweet corn, green beans, milkweed, and possibly black locust, sassafras, and black oak.

Continued stress from air pollutants on eastern white pine can be expected to cause changes in the ecosystem status of this forest species. Only 5.7% of the trees could be categorized as tolerant (0.4% had no visible symptoms). Berry and Hepting (1964) noted that 4.5% of their selected white pine clones were tolerant to air pollution. It is important to consider that (1) tolerant genotypes are not immune, and (2) forest tree species reduced in populations or biomass by air pollution are not necessarily replaced by other species of similar physiognomy. Selection against pollution-sensitive tree strains will not necessarily favor filling of those niches by air pollution-tolerant tree species (Farrar *et al*. 1977, Guderian 1977). Thus, the forest could slowly change towards a less diverse plant community.

CONCLUSIONS

Ozone data indicate that damaging concentrations occur over the entire ORBES region (See Figure V-4), as well as outside the region (Lioy and Sampson 1979). These concentrations are sufficient to cause slight to moderate foliar injury on sensitive tree species; in conjunction with elevated SO_2 levels (areas outlined in Figure IV-8), the O_3 concentrations can cause moderate to severe injury. In urban areas, trees are severely stressed by current air-pollution loads. Deleterious effects of air pollution are occurring in a wide variety of other sensitive plant species in the Ohio River Basin, including alfalfa, soybeans, tobacco, corn, and wheat (See Sections VII and VIII).

Classical air pollution symptoms were seen on 99.6% of the white pines sampled in Indiana during 1979 and 1980, and on 92.5% of the pines sampled in Wisconsin in 1980. Also, the presently occurring O_3 concentrations (alone and in conjunction with SO_2) are sufficient to cause alterations in normal physiological processes of white pines. Decreased photosynthesis and increased respiration result in a decrease in tree growth and vigor. Evidence is available suggesting that air pollution stressed trees are also more susceptible to other environmental stresses, such as fungal infections and insect infestations.

Ozone data from other states in the ORBES region suggest that white pines are being subjected to relatively uniform levels of O_3 exposure. Chronic exposure to presently occurring levels of O_3, and even higher levels in the future, could seriously affect the overall diversity and stability of the species making up the varied ecosystems in this region. With continued exposure to increasing concentrations of O_3 (in conjunction with other air pollutants), white pines and other O_3-sensitive species could be greatly reduced in this area. Mortality of oxidant-sensitive species is occurring in California (Miller 1973), and the reproduction of pines has been shown to be negatively influenced by oxidants (Houston and Dochinger 1977). If diversity declines in ecosystems with oxidant-sensitive species, the normal balance of the ecosystems will be altered. Consequently, other, natural perturbations (e.g., drought, insect infestations) will have a more severe effect on plant life in the affected areas.

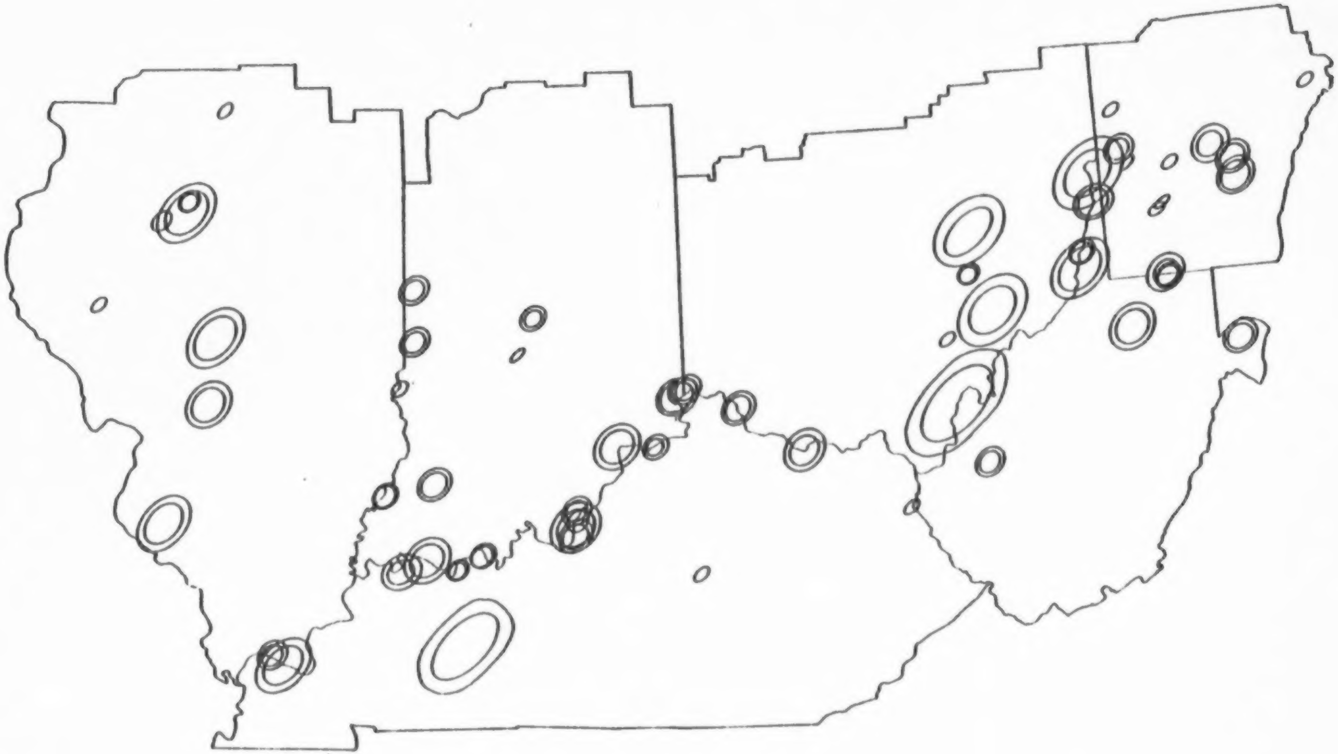

Figure IV-8. Areas around coal-fired power plants where calculated SO_2 concentrations are sufficient, in conjunction with regional ozone concentrations, to cause slight-to-moderate disease injury to tolerant white pines, and severe disease injury to sensitive white pines.

The index of foliar injury developed during this study appears to be an effective way of combining individual measures of plant condition to produce a composite index corresponding to the aggregate effect of exposure to air pollutants. This method, which can be used to compare the amount of pollutant-induced injury to pines at different locations, enables the use of white pine as a bioindicator of air pollution. Relative air pollutant levels can be interpreted from the index of injury values. However, if only the indices are examined, point source effects cannot be traced readily, because the widespread O_3 load seems to obscure the effects of local sources. Further analysis of the principal components of the index may enable differentiation between sites at a higher resolution.

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SECTION V

CALCULATION OF YIELD-LOSS COEFFICIENTS FOR MAJOR CROPS IN THE OHIO RIVER BASIN

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INTRODUCTION

Air pollution effects on crop yields constitute a problem of great practical importance, but considerable disagreement exists as to its magnitude. Difficulties in making damage assessments are caused, in part, by the fact that plants respond to the entire physical and chemical environment, not just to individual components. Since the environments of the laboratory and the field differ greatly, extrapolation from one to the other is often unwarranted. Ideally, prediction of effects of air pollutants upon field-grown crops should be based on data from experiments conducted with the proper crop, variety, and exposure regime at the site where the plants actually grow. Unfortunately, few data of this kind exist, requiring that assessments be based primarily on data from studies using controlled environments.

The purpose of this section is to present an analysis of the probable effects of air pollutants produced from fossil-fuel combustion upon yields of soybeans, corn, and wheat in the Ohio River Basin (ORB). The predicted yield reductions are based upon interpretation of reports published in the refereed literature plus some emerging results which have appeared in institutional reports and oral presentation abstracts. Data from research conducted in the field under conditions similar to those that occur in croplands have been given greatest emphasis. Where field data are lacking for a crop, results from laboratory or chamber studies were used, but shortcomings of the studies were considered in making the analysis.

Prediction of crop effects has been done cautiously in this section, based on established facts, yet recognizing new insights emerging from current research. The estimates of crop yield effects were generally conservative, that is, when data were conflicting, assumptions favored lower estimates of yield loss. In recognition of the difficulties involved in these predictions, anticipated effects are presented as a range, with maximum (upper bound), minimum (lower bound), and "most probable" crop loss coefficients.

REVIEW OF LITERATURE

Numerous studies have been made of the effects of air pollutants upon plant growth, and these have been summarized recently (U.S. EPA 1973, Mudd and Kozlowski 1975, Heck and Brandt 1977). At least 700 studies had been reported by 1973 (U.S. EPA 1973), but most of them were not conducted under conditions suitable for extrapolation to growing conditions in the Ohio River Basin (ORB).

Literature reviews show that most work has been conducted in greenhouse or growth chamber environments for short durations (commonly 1 to 8 hours). Evaluation of effects in most short-duration studies has been based on visible leaf injury, such as chlorosis or necrosis (Hill and Thomas 1933, Tingey *et al*. 1971a, Bennett *et al*. 1975.). In these studies, pollutant concentrations usually have ranged from 0.05 to 2.0 parts per million (ppm) sulfur dioxide (SO_2) or 0.05 to 1.0 ppm ozone (O_3). A second research approach has examined the inhibition of photosynthetic rates by exposure to air pollutants, frequently in the range of 0.2 to 5.0 ppm for 1 to 4-hour periods (Hill and Bennett 1970, Bennett and Hill 1973, Sij and Swanson 1974).

Both types of studies have shown that SO_2, O_3, nitrogen oxide (NO_2), and other pollutants that are primary or secondary products of fossil-fuel combustion can injure plants. Leaf lesions and reductions in photosynthesis, if they persist, can lead to crop damage (yield loss or reduction in crop quality), but there is little basis for assuming that crop damage from chronic exposure to pollutants can be predicted from leaf injury or short-term reduction in photosynthesis caused by acute injury. In fact, acute injury from SO_2 exposures for up to a few hours can often be distinguished visually from chronic injury caused by several days or weeks exposure (U.S. EPA 1973).

Yield Loss and Visible Leaf Injury

Some workers have concluded that visible injury must occur before yield reduction (Davis 1972, Heck and Brandt 1977). This conclusion is based on many, often early, reports of the response of various species to acute exposures (e.g., Hill and Thomas 1933, Katz 1949, Brisley *et al*. 1959).

Much recent evidence, however, particularly from chronic exposure studies, shows that yield or growth reduction in many plant species occurs in the absence of visible injury. Examples include the exposure of alfalfa to SO_2 and O_3 (Tingey and Reinert 1975), exposure of annual ryegrass and clover to O_3 (Bennett and Runeckles 1977), exposure of radish to SO_2 (Tingey *et al*. 1971b), exposure of soybeans to SO_2 (Sprugel *et al*. 1977, 1979), exposure of perennial ryegrass to SO_2 (Bleasdale 1952, Bell and Clough 1973), and examples of other species shown by other workers. Photosynthesis may also be inhibited without visible injury symptoms, as shown by Hill and Bennett (1970) with oats and alfalfa exposed to nitrogen oxides. Guderian (1977) has reviewed the many reactions at the cell and organ level that can occur without visible injury when plants are chronically exposed to low concentrations of pollutants. Since microscopic examination may reveal damage invisible to the naked eye (Mudd and Kozlowski 1975), inspection of plants without a microscope may not suffice as a method for detecting visible injury. Visibility of injury is apparently dependent upon growing conditions. Heagle (1979a) showed that changes in exposure regime alter the relative ranking of resistance of soybean cultivars to O_3, as measured by leaf injury. Growth and exposure temperatures can also influence foliar injury (Dunning *et al*. 1974).

Despite abundant evidence to the contrary, some workers have assumed that crop yield reduction should be suspected only in plants exhibiting visible injury symptoms in 5% or more of the total foliage blade area (Jones *et al*. 1979). Although it may hold under certain conditions, this assumption cannot be accepted as an unbiased criterion for identifying crop yield loss. The chief disadvantage of relying on visible injury as an indicator of yield or growth reduction is that in the absence of visible injury, changes in stomatal resistance, inhibition of photosynthesis, or reductions in growth or yield will be missed. If the visible effects criterion is adopted, evaluation of pollutant effects will be biased towards apparent resistance, even though deleterious effects may be occurring. A better approach would recognize that all plants with visible injury caused by air pollution stresses are exhibiting pollutant responses, but that not all pollutant effects are manifested as visible injury.

Environmental Conditions and Pollution Sensitivity

The environment of a growth chamber or greenhouse has been described as one which confers greater sensitivity to pollutants than do similar conditions in the field (U.S. EPA 1979). Experimental plants in controlled environments are typically well-watered, fertilized, exposed to high humidity (75%-85%), and grown at optimum temperatures. Under these conditions, stomatal resistance to the entrance of substances into leaves is reduced. In arid and semi-arid parts of the United States, where moist conditions do not occur regularly, stomatal resistance would probably be greater, reducing the efficiency of pollutant transfer into leaves. In most of the ORB, however, under typical conditions, the crop microclimate would appear to favor pollutant uptake. Mean daily relative humidity ranges from 70% to 80%, with the lowest relative humidity averaging 50% to 60% (Baldwin 1973). Within crop canopy, humidity would be higher. Precipitation, which averages 100 cm/yr or more over most of the region, eliminates the need for irrigation, and occasional dry periods generally occur in September after most crop growth has already occurred.

Wind is an aspect of gas transfer kinetics that has been given too little attention in the study of pollutant effects. Ashenden and Mansfield (1977) showed that at air velocities characteristic of closed chambers used in pollutant-effect studies, 0.11 ppm SO_2 can cause an apparent increase in growth of ryegrass, but increasing the air velocity results in about 25% reduction in growth. The wind speed associated with the growth reduction was less than that found in fields under "light breeze" conditions. The most reasonable explanation is that air movement reduces the thickness of the air layer immediately above leaf surfaces. In still air, this boundary layer can be depleted of gases, greatly reducing the transport of gas into the leaf (Larcher 1975). The well-known role of wind in reducing the resistance to carbon dioxide (CO_2) absorption by leaves applies also to water vapor and other gases (Holmgren et al. 1965, Wuenscher and Kozlowski 1971, Unsworth et al. 1976).

As pointed out by Ashenden and Mansfield (1977), results of many studies on plant response to air pollutants can no longer be considered valid for estimating effects on crops growing in the field, because sufficient air movement was not provided. In these studies the amount of pollutant entering a leaf probably was greatly reduced by the presence of a thick boundary layer. In the Ashenden and Mansfield study, apparent growth resulted from a smaller rate absorption of a pollutant than would occur at the same ambient pollutant concentrations associated with a higher air velocity.

The same criticism may apply to the use of open-topped field chambers (Haegle et al. 1979a), although they represent a great advancement over conventional growth chambers. Air velocity in open-topped field chambers has been reported to vary from 1.47 to 2.92 kilometers per hour (km h^{-1}) (Heagle et al. 1979a), but wind velocity in alfalfa and corn fields often exceeds 7 km h^{-1} on clear midsummer days (Lemon 1963). The windspeed designated by meteorologists as a "light breeze" is commonly exceeded in the field. It includes speeds of 5.4 km h^{-1}, which Ashenden and Mansfield (1977) measured 20 cm above the ground. Because of the structure of crop canopies, air velocity may vary greatly along a vertical profile. As a result, boundary layer conditions may also vary from leaf to leaf in a manner which is quite different in field and chamber environments. Another problem with open-top field chambers is the intrusion of ambient air through the top of the chamber (McCune et al. 1976c) and degradation of O_3 in the duct system (Heagle et al. 1979a). The significance of these environmental modifications is unknown, but they emphasize the need to be cautious in applying research results from chamber studies to field stations.

Interacting Effects of More Than One Pollutant

Near ORB power plants, O_3 and SO_2 and possibly other pollutants (e.g. nitrogen oxides) occur at potentially damaging levels. Increasing evidence suggests that the presence of two or more pollutants in a crop environment may cause additive or synergistic effects. Additive, less than additive, and greater than additive effects have been observed in laboratory experiments at specific concentrations for some species (Tingey and Reinert 1975). Other studies report consistent decreases in growth caused by pollutant mixes, which exceeded damage caused by single pollutants (Tingey et al. 1973b, Heagle et al. 1974, Guderian 1977). Because crops are exposed to variations in all environmental factors, including pollutant concentrations, additive or synergistic effects would be expected at certain times. The presence of NO_2 near ORB power plants, sometimes at concentrations of 0.05 ppm or more, may also cause injury, particularly in the presence of SO_2 (Tingey et al. 1971a).

More data are needed, particularly under field conditions, on the effects of pollutant mixes at realistic concentrations and during sensitive stages of the crop life cycle. Nevertheless, the weight of available evidence suggests that substantial effects from present pollutant mixes should be expected (Guderian 1977). Evidence does not support the notion that inadvertent reduction of air pollution problems may occur due to a cancelling out of effects between pollutants. Thus pollutant interactions might be expected to increase crop losses above those presented in this analysis.

The Lower Limit for SO_2 Exposures Causing Crop Yield Reductions

One of the most difficult questions in pollutant effects research is determination of a gaseous concentration below which--in the presence of slightly elevated O_3 or nitrogen oxides (NO_x) concentrations--there may be no significant additional effect from the toxicant on the most sensitive commercial species.

Guderian (1977) questions the idea of a "threshold of injury" for certain species. In fact, for some substances there may be no exposure below which an effect (that could be considered deleterious) does not occur. Instead, apparent thresholds may only be artifacts of the measurement system employed.

For example, 0.01 ppm SO_2 for 10 minutes can change the diffusive resistance of stomata (Montana Air Quality Bureau 1979), and lowered diffusive resistance would permit entry of other pollutants. Certain experimental approaches would miss these effects because the effect of only one pollutant was considered, or because effects at low concentrations were not carefully sought. Evidence reviewed in previous sections indicates significant negative physiological effects can result from exposures to SO_2 in the few-hundred micrograms-per-cubic-meter range. Other work has shown that recovery from these effects may take several days to 2 weeks, but total recovery may not occur (Bennett and Hill 1973). For these reasons, and recognizing the logarithmic distribution of SO_2 exposures, 1-hour exposures occurring twice a month would be a measure of potential interest in evaluating very sensitive species such as eastern white pine or alfalfa. A later section explores in more detail the use of the monthly second-highest hourly SO_2 observations for characterizing SO_2 exposures around existing power plants.

The effects reviewed here and in previous sections indicate that hourly exposures, twice per month, of 200 to 300 micrograms/cubic meter ($\mu g/m^3$) under O_3 concentrations of 0.05 to 0.10 ppm may reduce growth or yield of sensitive species. The Montana Air Quality Bureau (1979) summarized the SO_2 effects literature and found 24 published papers that report deleterious effects in a wide range of species from exposure to concentrations at or below 0.05 ppm SO_2. Most of these studies were not designed to determine effects upon yield or growth. But large yield and growth reductions in ryegrass have been found to result from exposure to 0.037 ppm SO_2 over long periods (Crittenden and Read 1978). Since the frequency and duration of exposure in the Crittenden and Read work was much higher than in the ORB areas defined by a 0.05 ppm exposure zone, unrealistically high losses might be projected from their experimental results. Nevertheless, levels of SO_2 in the 0.025-0.05 ppm range must be considered high enough to reduce growth and yield of sensitive species, particularly in the presence of other phytotoxic gases.

Cumulative Exposure to O_3 and SO_2 as a Predictor of Crop Losses

To assess crop yield losses from low-level chronic exposure to O_3 and SO_2, concentrations and duration of total exposure must be considered. Mathematical equations describing visible injury in response to short exposures fit existing data well and can be used to predict visible leaf surface injury, if concentration and duration of exposure are known (U.S. EPA 1973). Other mathematical models can successfully predict leaf surface injury based on an O_3 concentration and exposure duration over 8 hours (Larsen and Heck 1976, Nouchi and Aoki 1979). However, these models were not designed for predicting yields after chronic exposures and thus are of little value in predicting losses resulting from a season-long dose.

Calculation of losses from chronic exposure involves many interacting factors, including seasonal growing conditions, variations in pollutant exposure regimes, and sensitivity of different phenophases. Thus far, the best attempts to interpret chronic exposure effects have come from Heagle *et al*. (1972, 1974, 1979b, 1979c) who treated plants with single O_3 concentrations for long durations in closed-top or in open-top field chambers. Effects of chronic SO_2 exposure on soybean yield have also been studied by Heagle *et al*. (1974) using field chambers, by Miller *et al*. (1977, 1979), and by Sprugel *et al*. (1977, 1979) using a field fumigation exposure system. Only in the latter study was the artificial environment formed by enclosures eliminated; SO_2 was administered to field plants in an exposure regime approximating actual fumigation dynamics.

Continuous or daily pollutant exposures used by experimenters (e.g., Tingey *et al*. 1973b, Heagle *et al*. 1974, 1979b) do not allow for plant recovery, which has been demonstrated over short durations (Guderian 1977). Although no data exist on whether plants recover over an entire growing season with intermittent or variable exposure to a toxin, recovery may be expected at least during vegetative growth stages, depending on exposure duration, total dose, and period between exposures. Using cumulative exposures calculated from continuous exposure studies could produce inflated estimates of yield loss, if the potential for recovery is not considered. Unfortunately, little evidence is available to illuminate the recovery problem. Some information from short-term exposures suggests, however, that a growth loss may occur in kidney beans when exposures are spaced apart by as much as a week (McLaughlin *et al*. 1979). In the eastern United States, the possibility of recovery from damage caused by O_3 at concentrations between 0.05 to 0.10 ppm may be low because prolonged exposures within this range are becoming common (Heagle *et al*. 1979c, Angus and Martinez 1975).

Under conditions of prolonged exposure to high levels of O_3, plant injury or damage can be expressed as a function of the cumulative exposure over a season. Kelleher and Feder (1978) found that cumulative leaf damage in tobacco was a function of the number of times that O_3 exceeded 0.04 ppm for more than 4 hours. Alfalfa yield loss and defoliation were found to be related to cumulative O_3 exposure along an ambient gradient in southern California by Oshima et al. (1976). Alfalfa damage could be predicted from the cumulative O_3 dose exceeding 0.10 ppm (and lower concentrations) over a 7-month period.

The cumulative exposure approach also can be used in assessing SO_2 damage, although the high peaks characteristic of SO_2 fumigations may cause acute injury that cannot be estimated by summing hourly values of SO_2 exposure. Sprugel et al. (1979) found that intermittent exposure to SO_2 at concentrations of 0.09 to 0.79 ppm over a 2-month period caused yield loss in soybeans that correlated with the cumulative SO_2 exposure. Within or near this concentration range, similar cumulative exposures may be expected to have similar effects.

An assessment based on cumulative exposure also was used for predicting yield loss from long-term exposure to elevated O_3 concentrations in the Ohio River Basin. The cumulative exposure approach reflects increasing recognition that chronic exposures to low pollutant concentrations, rather than acute exposures, are responsible for much yield loss and decline in vigor (Feder 1973). The estimation of crop loss coefficients in this analysis, therefore, is based on the total exposure to SO_2 or O_3 in the Ohio River Basin, expressed as $\mu g \cdot m^{-3}h$ for each pollutant.*

Crop yield loss functions have been expressed as a range in values, reflecting the range of factors influencing crop response. Growing conditions, sensitivity of life-cycle stages, cropping conditions and choice of cultivars, are principal variables that will interact to determine the actual crop response. Presentation of results as minimum, maximum and probable loss functions is intended to accomodate the range of actual crop responses over the Ohio River Basin region to pollution conditions observed in 1977. Loss functions are summarized in Table V-6 at the end of the chapter.

Product quality may also be affected by low levels of air pollutants, but this phenomenon has not been incorporated into crop loss estimates. Many aspects of product quality, such as appearance and texture, are not readily quantified, but some observers have noted that they accompany yield reduction (Heagle et al. 1972). In a complete analysis, even if loss in product quality occurs without yield reduction, economic losses may still occur, and these must be attributed to the costs of air pollution.

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* Micrograms per cubic meter/hour. At 25°C and atmospheric pressure of 760 mm Hg, a 1-hour exposure at 1 ppm $SO_2 = 2620\ \mu g \cdot m^{-3}h$, and a 1-hour exposure at 1 ppm $O_3 = 1960\ \mu g \cdot m^{-3}h$.

CALCULATION OF CROP LOSS COEFFICIENTS FROM O_3 IN THE OHIO RIVER BASIN

In contrast to SO_2 exposure effects usually found in the vicinity of major point sources, O_3 effects have been observed to be a region-wide phenomenon. The elevated ozone clouds in the Ohio Basin generally occur over large rural areas far from sources of possible precursors. Thus, effects of O_3 can occur where SO_2 and NO_x concentrations are low. Interactions with primary emissions from power plants appear not to be important except on a local scale near point sources. Few data are available, however, to evaluate the possibility of interactions between O_3 and the widespread occurrence of acid deposition. Generally, the O_3 loss functions presented below reflect the effects of O_3 in the absence of damaging levels of other pollutants.

In developing an estimator of prospective field crop reductions due to gaseous pollutants (O_3 or SO_2), two types of data are available and must be used conjunctively:

(1) Experimentally determined reductions in yield (expressed as a % departure from control yields) for the known pollutant exposures (concentration x hours);

(2) Known or locally interpolated exposures of the crops to O_3 (or SO_2) in the field environment during the growing season.

If a generally consistent relationship can be shown between the experimentally observed yield reductions, and observed cumulative O_3 exposures during the experiments, and if most of the experimental conditions are representative of the humid Ohio Valley area, then an ORBES region "crop loss coefficient" can be interpolated from the curve using ORBES area O_3 exposures. The following subsections explore the data base for these relationships for three major crop species, soybeans, corn and wheat. Generally, the O_3 loss functions presented below reflect effects of O_3 in the absence of damaging levels of other pollutants.

Estimating the Effects of Cumulative O_3 Exposure on Soybean Yield

In the Ohio River Basin, crops are exposed for extended periods to elevated concentrations of O_3, maximum hourly O_3 values frequently exceed 0.12 ppm at both urban and rural monitoring stations (Figure V-1). Evidence that O_3 levels in rural areas exceed urban values in the eastern United States (Angus and Martinez 1975, Isaksen *et al*. 1978) suggests that ORB croplands are exposed to high maximum levels. This pattern of rural enhancement of O_3 also holds for the Indianapolis area, where outlying stations frequently register higher O_3 levels than central urban stations (Haering 1977). One mechanism by which O_3 enhancement can occur has been described for air parcels over the Great Lakes (Miller *et al*. 1978).

Estimation of the ORB seasonal O_3 burdens utilized the relationship between peak and mean concentrations commonly observed where there is continuous monitoring of O_3. Mean hourly values during the period 0930-

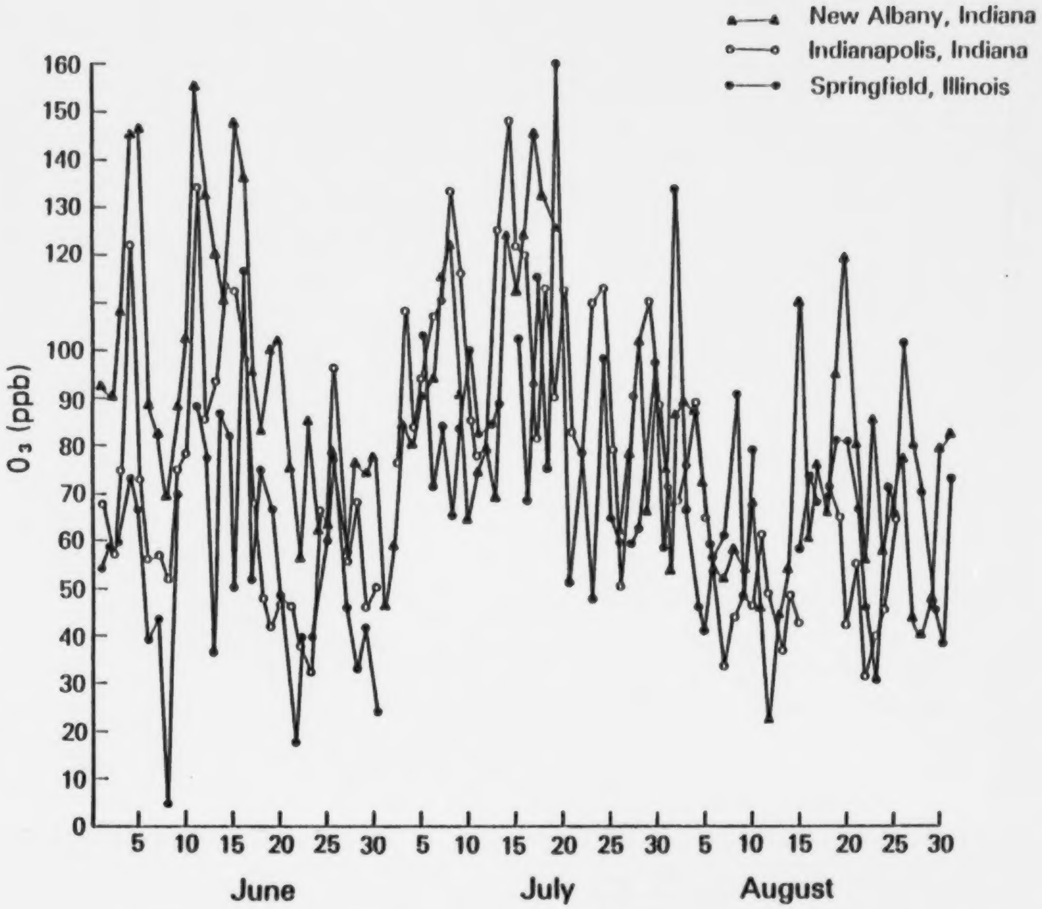

Figure V-1. Maximum hourly ozone averages for June, July, and August 1977 at three urban stations in the Ohio River Basin. Data are from Teknekron Research, Inc.

1630 often are at least 70% of the peak hourly concentration (Figure V-2). Mean hourly exposures at ORBES area monitoring sites, therefore, were estimated by multiplying 0.7 times the hourly maximum during the 7-hour daily period. The period 0930 to 1630 hours was selected, since during this period plants are assumed to be most sensitive to pollutant effects, as suggested by Heagle et al. (1979). If this assumption were in error, the crop loss would be underestimated because it implies that exposure to elevated O_3 during the other 17 hours of each day causes no damage. The cumulative exposure during the sensitive period has been calculated by multiplying the mean hourly exposure by 7.

The total O_3 exposures in the Ohio River Basin have been reduced by subtracting background O_3 exposure equivalent to 0.03 ppm (59 $\mu g \cdot m^{-3}$) for the 7-hour period. The same correction was applied to the experimental studies. Ozone concentrations above background appears to result largely from anthropogenic sources to which plants have been exposed only over the past few decades. In contrast, a background concentration in the Antarctic averages 0.02 to 0.03 ppm (Altshuller and Cadle 1971), a level which presumably has been a natural constituent of the lower atmosphere during long periods of the earth's history and to which evolutionary adaptation by plants has occurred. Although plants respond physiologically to O_3 regardless of its source, loss functions were calculated only from cumulative exposure above background, reflecting the experimentally demonstrated inability of plants to accomodate ambient O_3 concentrations exceeding background levels.

Several studies have considered effects of chronic O_3 exposure upon soybean growth and yield (Table V-1). Observed effects vary, although it is certain that chronic exposure to elevated O_3 levels for time periods that approach the length of the crop growing season have detrimental effects. Heggestad et al. (1977) have shown that soybean yield is depressed 20% by ambient O_3 levels characteristic of eastern Maryland, and Heagle et al. (1979c) have shown similar yield depressions for North Carolina. Oshima et al. (1976) demonstrated that ambient O_3 in southern California causes severe damage to alfalfa in proportion to cumulative exposure.

Heggestad et al. (1977) reported results of a 3-year study of yield of four soybean varieties exposed to ambient O_3. During this period oxidants rarely reached an hourly average of 0.15 ppm, and exceeded an hourly average of 0.10 ppm on only a few days each season (Heggestad 1980). Plants growing in large open-top chambers in the field or plants in chambers with carbon-filtered air served as controls. Reductions in yield were 29, 9 and 17% (mean loss = 18%) in the years 1973 to 1975 (Table V-1). Over the same period, yields in untreated plots varied by 23% from the 3-year mean, underscoring the need for multi-year studies which can integrate differences in cropping practice and microclimate. Ozone data were reported daily for 11-hour periods in 1975.

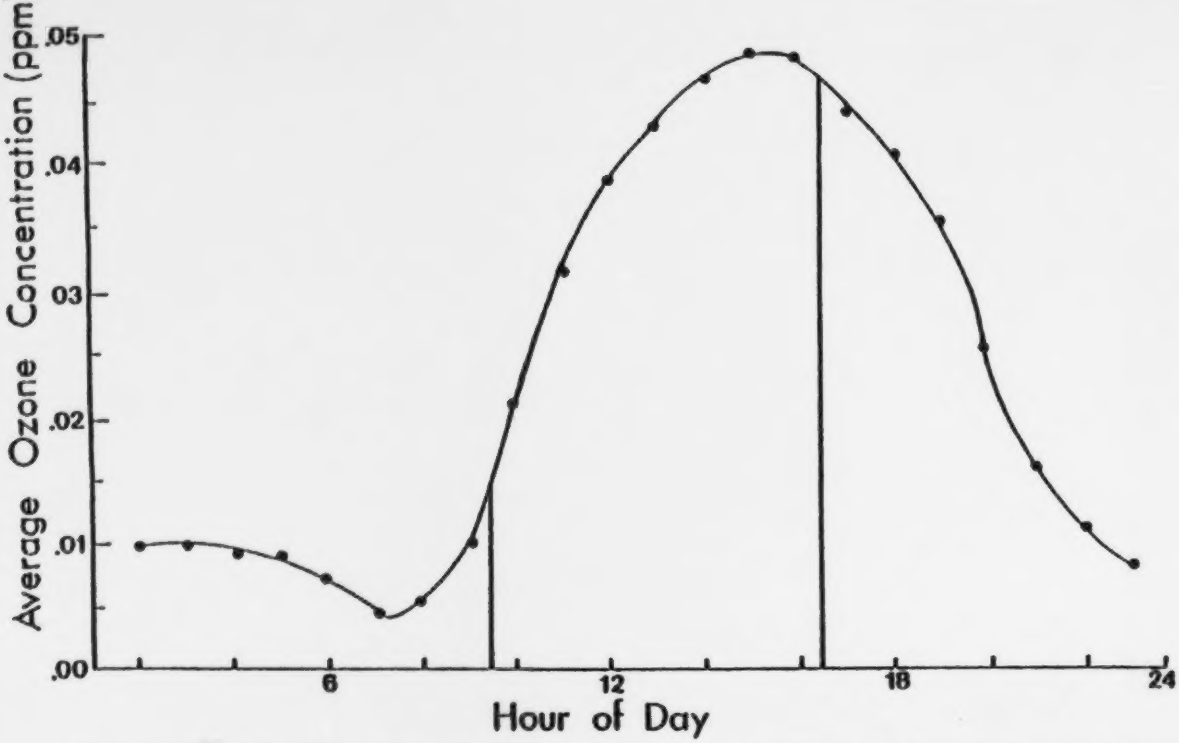

Figure V-2. Profile of daily ambient ozone at Indianapolis (top) and Raleigh, North Carolina (bottom). Indianapolis data are mean weekday values for summer 1974 (Lovelace *et al.* undated); Raleigh data are for April 9 to May 31, 1977, from Heagle *et al.* (1979c). The ratio between the mean hourly value and the peak value during the 0930-1600 period is 0.78 for the Indianapolis data, and 0.85 for the Raleigh data. Note that O_3 levels near the maximum occur during daylight hours outside the period 0930-1630.

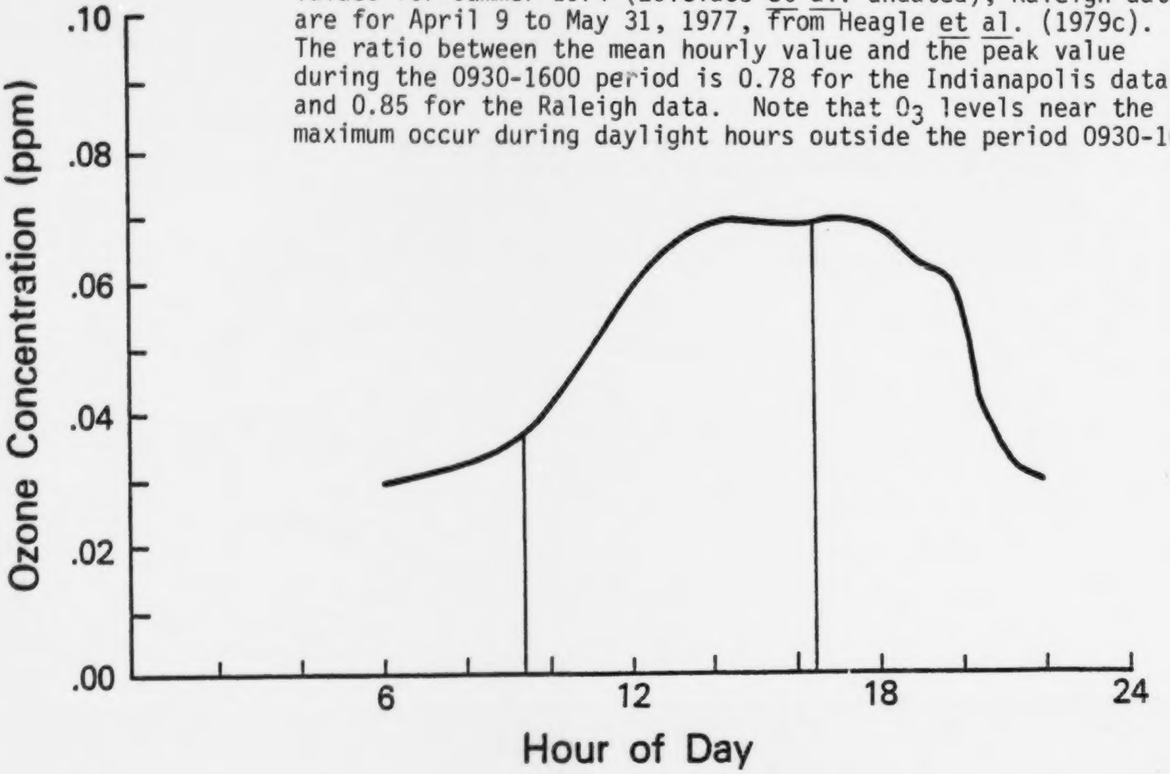

TABLE V-1. EFFECTS OF CHRONIC OZONE EXPOSURE UPON SOYBEAN YIELD. DATA ARE FROM CHRONIC EXPOSURE STUDIES USING CONCENTRATIONS WHICH HAVE BEEN MEASURED IN THE OHIO RIVER BASIN.

Days	Hours	Concentration (ppm)	Cumulative Exposure[a] (mg·m^{-3}h)	Effect
1. Heagle *et al*. (1974)				Compared to controls in ambient air:
43	258	0.10	35.4	66% reduction in plant fresh-weight
92	552	0.10	75.7	50% reduction in pod fresh-weight; 55% reduction in plant fresh-weight
133	798	0.10	110.0	63% reduction in seed harvest; 66% reduction in plant fresh-weight
43	258	0.05	11.8	27% reduction in plant fresh-weight[b]
92	552	0.05	25.5	25% reduction in pod fresh-weight[b]; 25% reduction in plant fresh-weight[b]
133	798	0.05	36.3	20.5% reduction in seed harvest[b]; 24% reduction in plant fresh-weight
In presence of 0.10 ppm SO_2:				
43	258	0.10	35.4	54% reduction in plant fresh-weight[b]
92	552	0.10	75.7	57% reduction in pod fresh-weight; 54% reduction in plant fresh-weight
133	798	0.10	110.0	70% reduction in seed harvest; 73% reduction in plant fresh-weight
2. Heagle *et al*. (1979c)				Compared to controls in filtered air:
	630	0.07	49.4	6% yield reduction
	630	0.11	98.8	31% yield reduction
	630	0.15	148.0	53% yield reduction
3. Tingey *et al*. (1973b)				Compared to controls in filtered air:
15	120	0.10	16.5	15% dry-weight reduction; 22% fresh-weight reduction
15	120	0.05	4.7	4.6% dry-weight increase[b]; 2% fresh-weight decrease[b]
In presence of 0.05 ppm SO_2:				
15	120	0.05	4.7	5.4% increase in shoot dry-weight; 20% reduction in root dry-weight; 14% reduction in total fresh-weight
4. Heggestad *et al*. (1977)				
62	1488	ambient	34[c]	29.3% reduction in yield
62	1488	ambient	34	8.7% reduction in yield
62	1488	ambient	34	17.2% reduction in yield

[a] Above background of 59 μg·m^{-3} (0.03 ppm)

[b] Statistically non-significant for experimental conditions used ($P = 0.05$)

[c] Exposure calculated for 12-hour daytime period from frequency and summary tables.

Since cumulative O_3 exposure or daily O_3 values were not presented by Heggestad et al., cumulative exposure was calculated from mean hourly oxidant values for each month (over 24 hours) and from frequency distributions of mean hourly oxidants values at Beltsville, Maryland in the vicinity of the study plots (Heggestad et al. 1977). The estimated cumulative exposure for all 11-hour days (0900 to 2000) in July and August 1975, the year when the intermediate yield loss was observed, was 34,100 $\mu g \cdot m^{-3}h$. This figure was used to characterize the 1973 and 1974 exposures as well. Although the actual value may differ, if only the 7-hour daytime exposure is of principal importance in influencing plant response, the response/exposure function has been conservatively calculated. The study demonstrates that ambient O_3 levels characteristic of the eastern U.S. can depress soybean yields nearly 20% over several years. Since these O_3 levels frequently occur throughout large areas of the eastern United States (Angus and Martinez 1975), soybean yield depression probably is widespread.

Calculation of O_3 Loss Coefficient for Soybean Yield

The relationship for reduction of soybean yield from O_3 exposure has been developed from regression curves fitting experimental data on yield to the logarithmic transformation of cumulative O_3 exposure above background (Figure V-3). Ozone exposure data are presented for four stations during the June through August 1977 period (Table V-2). Cumulative O_3 exposures above background ranged from 22,600 $\mu g \cdot m^{-3}h$ at Springfield to 37,640 $\mu g \cdot m^{-3}h$ at New Albany. The mean of these exposures (30,100 $\mu g \cdot m^{-3}h$) is similar to values for Youngstown and Indianapolis, and may represent an intermediate value for the region as a whole. The range of exposures over the Ohio River Basin as indicated by the data from the four widely separated stations, suggests that in spite of variation, a regional blanket of O_3, locally enhanced, appears to be characteristic during summer months. In 1978, sixteen monitoring stations in Illinois recorded more hours with O_3 levels exceeding 0.08 ppm than did the Springfield station according to the Illinois Environmental Protection Agency (1978), suggesting that many Illinois crop areas are exposed to O_3 levels higher than those recorded at Springfield. Thus exposures noted at Springfield may represent the low end of the O_3 exposure scale in the region, while observations at New Albany typify near maximum exposures.

The best fit of statistically significant yield data is obtained from Heagle et al. (1974) and Heggestad et al. (1977), summarized as Curve I, Figure 3, which suggests a maximum loss coefficient of 26% for the mean cumulative exposures at the Springfield and New Albany stations. Similar loss coefficients are suggested from Curve III which incorporates data from Heagle et al. (1979c). These results are influenced by the possible outlier at the coordinates (35,000, 66). A regression made by deleting this point from the data associated with curve I produces Curve II with a higher coefficient of determination (r^2 = .92) suggesting a loss coefficient of 14% for the mean O_3 exposures at Springfield and Albany. A similar improvement in r^2 results from deleting this point from Curve III data, producing a mean loss of 12%, ranging to over 19% where exposures

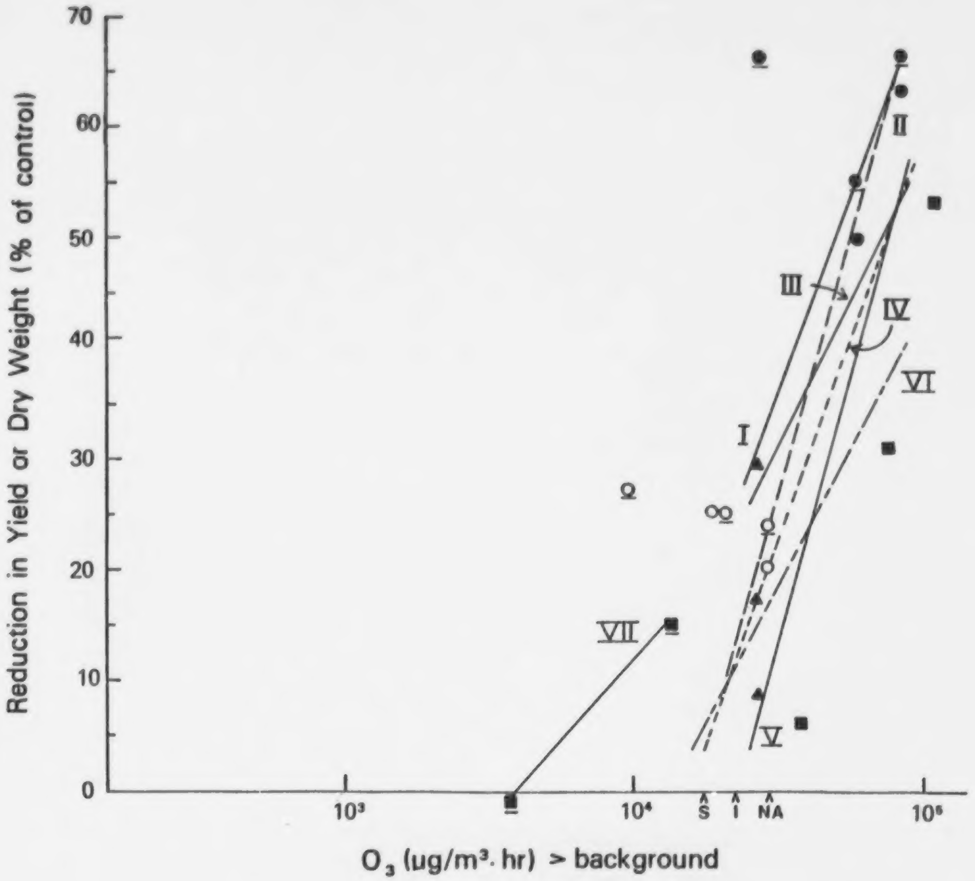

Figure V-3. Effects of chronic ozone exposure upon soybean yield (seed or pod weight). Data are from Heagle et al. (1974) (o=yield) (o=plant weight, ●=non-significant data at $P=0.05$); Heggestad et al. (1977) (▲); Heagle et al. (1979) (■); and Tingey et al. (1973b) (■= plant weight). Ozone exposures from Heggestad et al. (1977) were calculated from frequency distributions and average oxidant values. Curve I is a least-squares regression fit of all statistically significant yield data from Heagle et al. (1974) and Heggestad et al. (1977). The equation is $Y=68.572 \text{ Log } X-281.218$ ($r^2=0.494$; $0.1>P>0.05$). Curve II is a fit of the same data without the apparent outlier at coordinates (66, 35, 400). The equation is $Y=92.325 \text{ Log } X-399.714$ ($r^2=0.923$; $P<0.001$). Curve III is a fit of Heggestad et al. (1977) and Heagle et al. (1979, 1974) without the apparent outlier. The equation is $Y=73.778 \text{ Log } X-318.407$ ($r^2=0.629$; $P<0.01$). Curve IV is a fit of the same data including the outlier. The equation is $Y=52.9214 \text{ Log } X-213.8828$ ($r^2=0.319$; $0.1>P>0.05$). Curve V is a fix of data from Heagle et al. (1979, 1974) without the outlier. The equation is $Y=95.081 \text{ Log } X-425.029$ ($r^2=0.485$; $0.1>P>0.05$). Curve VI is a fix of data from Heggestad et al. (1977) and Heagle et al. (1979). The equation is $Y=48.7465 \text{ Log } X-206.9971$ ($r^2=0.594$; $0.1>P>0.05$). Curve VII connects the plant weight data of Tingey et al.(1973b). The data for the curves are given in Table V-1.

TABLE V-2. CUMULATIVE EXPOSURE TO SO_2 AND O_3 AT MONITORING STATIONS IN THE OHIO RIVER BASIN.

Monitoring Station	Cumulative Exposure ($mg \cdot m^{-3}h$)						
	SO_2 (1977)				Ozone[a] (1977)		
	June-Aug	July-Aug		Winter[c]	June-Aug	July-Aug	Winter
		Total	>0.1ppm				
Clifty Creek Power Plant (0.2 ppm impact zone)	105.8	81.8	27.4	104.0			
New Albany, Indiana					37.6	25.1[b]	23.2[d]
Springfield, Illinois					22.6		
Indianapolis, Indiana					28.0		
Youngstown, Ohio					27.9		

[a] Above background of 59 $\mu g \cdot m^{-3}h^{-1}$ for a 7-hour day

[b] Calculated as 0.667 of the June-August exposure

[c] October, November, March-June
Calculated as follows: (June and October) = 2 x ($\bar{x}$ June 1974 and June 1977); other months = 0.75 x June

[d] Assumes June, September and October = 3 x lowest summer month; other months ignored

reach those of New Albany (Curve IV - Figure V-3). If the loss coefficient is based on exposures weighted downwards slightly to reflect those observed at Springfield, losses would be to 2 to 3% lower, judging from Curves II and IV, but the presence of higher O_3 exposures over many areas in Illinois, plus interpretation from Curves I and III, suggests the minimum loss function for the region as a whole should be at least 5%. Only Curve V predicts lower loss functions for much of the region, although even those data suggest a 10% loss for New Albany exposures.

Further evidence of a minimum of at least 5% over the region is suggested by the mean exposure for Springfield and Youngstown (25,150 $\mu g \cdot m^{-3}h$), the two lowest of the four reported stations. The estimated loss coefficient at this exposure is 6 to 7% from the two equations with statistically significant (at $P = 0.05$) correlation coefficients. Curve VI supports this interpretation. Thus, based on the equations with highest statistical significance, and the oxidant loadings observed over the region, a minimum loss function of 5%, a "most" probable loss function of 14%, and a maximum loss function of 26% appear to reflect the range of effects on soybean yields associated with O_3 conditions observed in 1977.

Calculating loss functions for exposures recorded at Springfield, Indianapolis and Youngstown monitoring stations from linear regressions I through V requires extrapolation below the lowest cumulative doses covered by regressions I and II (34,100 to 49,400 $\mu g \cdot m^{-3}h$). Several lines of evidence, however, indicate that yield reductions from lower exposures are probable, thus supporting the extrapolations. First, there is only a small difference between the observed cumulative exposures and those reported in the literature to cause damage. Since there is no evidence that a sudden threshold would develop at slightly lower exposures than are reported in the experiments, some crop damage at O_3 levels only 10 to 20% below the experimental levels is likely.

Secondly, evidence for substantial damage to soybeans at the reported ozone levels is shown by the 15% reduction in dry weight in pre-flowering plants found by Tingey _et al_. from 16,500 $\mu g \cdot m^{-3}h$ of ozone administered at 0.10 ppm. Both this concentration and the cumulative exposure were reached or exceeded at the ORB monitoring stations. Also, the 25 to 27% reduction in fresh weight of plants and pods found at cumulative exposures ranging from 11,800 to 36,300 $\mu g \cdot m^{-3}h$ by Heagle _et al_. (1974), although reported as not statistically significant (at $P = 0.05$), indicates that damage may occur at these lower levels. Damage also should be expected because many summer days at Indianapolis and Youngstown are characterized by peak hourly averages at or above 0.10 ppm (Table V-2), concentrations commonly associated with acute and chronic damage to a wide variety of plants, many of them considered more resistant to ozone than soybeans.

Calculation of O_3 Loss Coefficient for Corn

Evidence that cumulative exposures of 30,000 to 40,000 $\mu g \cdot m^{-3}h$ O_3 can cause significant yield loss was presented by Oshima (1973) who observed a 12.6% yield reduction of Golden Jubilee sweet corn exposed to 0.20 ppm O_3

for 9 hours per week (3 hours per day for 3 days per week), equivalent to a 33,000 μg·m^{-3}h cumulative exposure. Further evidence comes from Heagle et al. (1972) who reported that an 8.7% reduction yield of Golden Midget sweet corn resulted from exposure to 15,800 μg·m^{-3}h given as 0.05 ppm O_3 for 6 hours daily. In addition, yield was reduced by nearly one-half by exposure to 55,200 μg·m^{-3}h given as 0.10 ppm O_3 from seeding to maturity (Table V-3). Thus, the data show substantial yield losses can occur at a mean hourly concentration near 0.05 ppm such as commonly occurs in the ORB.

The first results of a field study of the effects of chronic O_3 exposures upon yield (kernel weight per plant) of field corn were published in 1979 (Heagle et al. 1979b). Results show ambient O_3 that averaged 0.06 ppm (cumulative exposure = 36,700 μg·m^{-3}h) during the 7-hour daytime period caused a reduction in kernel dry-weight of 7.4% in Coker 16 variety (Table V-3). The control plants for this experiment were exposed to charcoal-filtered air averaging 0.02 ppm O_3 in an open-top field chamber described elsewhere (Heagle et al. 1979b). Plants exposed to ambient air were not enclosed. When the same controls were compared to plants exposed to 0.07 ppm O_3 in chambers, no yield reduction was observed. The authors do not explain the apparent discrepancy in the responses to ambient and chamber regimes, but possibly O_3 exposure beyond the 7-hour daytime period or the presence of other pollutants affected the plants exposed to ambient air.

Corn varieties apparently differ importantly in sensitivity to O_3 (Heagle et al. 1972, 1979b). For example, exposures causing large yield losses in the variety Golden Midget caused smaller, non-statistically significant losses in percent ear-fill, number of kernels/plant and kernel dry weight in the variety White Midget. These data were not considered in making loss estimates because midget varieties are not important crops in the Ohio Basin, but the three parameters describe different aspects of product quality, and even small losses could be economically significant to farmers. Clearly, the varietal differences in O_3 sensitivity are a primary factor in determining yield loss, and thus actual crop damage will vary with varieties planted. The presentation of yield loss functions as a range would allow for some of the possible varietal differences in yield.

A curve connecting the data available for sensitive varieties (Figure V-4, Curve I) from Oshima (1973) and Heagle et al. (1979b) suggests a loss function within the range of 6% to 15% for exposures over the period June through August at Springfield and New Albany. If exposures only during July and August affect yield, losses would range from 6 to 7%. For resistant varieties, data from Thompson et al. (1976) and Heagle et al. (1979b) suggest loss extremes over all varieties for both locations vary from 5 to 8% (Curve II). The intermediate value for the mean exposure characteristic of Springfield and New Albany is 8%, which appears to be "most probable" loss function. The value is close to the average yield reduction of 9% observed in four sweet corn varieties in 1977 and 1978 in Maryland under similar ozone conditions (Heggestad 1980).

TABLE V-3. EFFECTS OF CHRONIC OZONE EXPOSURE UPON YIELD OR GROWTH OF *Zea maize* (corn) *Triticum sativum* (wheat), AND *Lolium multiflorum* (ryegrass)

Days	Total Hours	Concentration (ppm)	Cumulative Exposure[a] (mg·m^{-3}h)	Effect
1. Oshima (1973) - Corn (Golden Jubilee)				
77[b,c]	99(3)[e]	0.20	33.0	12.6% reduction in kernel dry weight
77[b,c]	99(3)	0.35	62.1	22.4% reduction in kernel dry weight
2. Heagle *et al*. (1972) - Corn (Golden Midget)				
67	402(6)	0.05	15.8	8.7% reduction in kernel dry weight[d]
67	402(6)	0.10	55.2	45% reduction in kernel dry weight
3. Heagle *et al*. (1979b) - Corn (Coker 16)				Compared to charcoal filtered air control:
88	616(7)	0.06	36.2	7.4% reduction in kernel dry weight (ambient air)
88	616(7)	0.07	48.2	2.8% increase in kernel dry weight[d]
88	616(7)	0.11	96.6	3.9% decrease in kernel dry weight[d]
88	616(7)	0.15	145.0	15.7% decrease in kernel dry weight
4. Thompson, *et al*. (1976)				
79	1185(15)	ambient	178b	23.1% kernel fresh-weight reduction of primary to secondary ears in variety Monarchy
79	1185(15)	ambient	178b	13.6% kernel fresh-weight reduction of primary and terminal ears of variety Bonanza
5. Heagle *et al*. (1979c) - Winter Wheat (4 cultivars)[f]				
	(7)	0.06		3% yield reduction for plants in the ground
		0.10		16% yield reduction for plants in the ground
		0.13		33% yield reduction for plants in the ground
6. Bennett and Runeckles (1977) - Ryegrass (*Lolium multiflorum*)				
48	384(8)	0.03	15.0	5.1% reduction in plant shoot dry weight[d]
48	384	0.09	45.2	22.2% reduction in plant shoot dry weight

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[a] Above background of 0.03 ppm (except 0.03 ppm of Bennett and Runeckles)
[b] Estimated; actual value not reported
[c] Three fumigations per week
[d] Statistically non-significant at 5% probability level
[e] Exposure in hours per day are in parentheses
[f] Length of exposure and time of year of experiments not reported

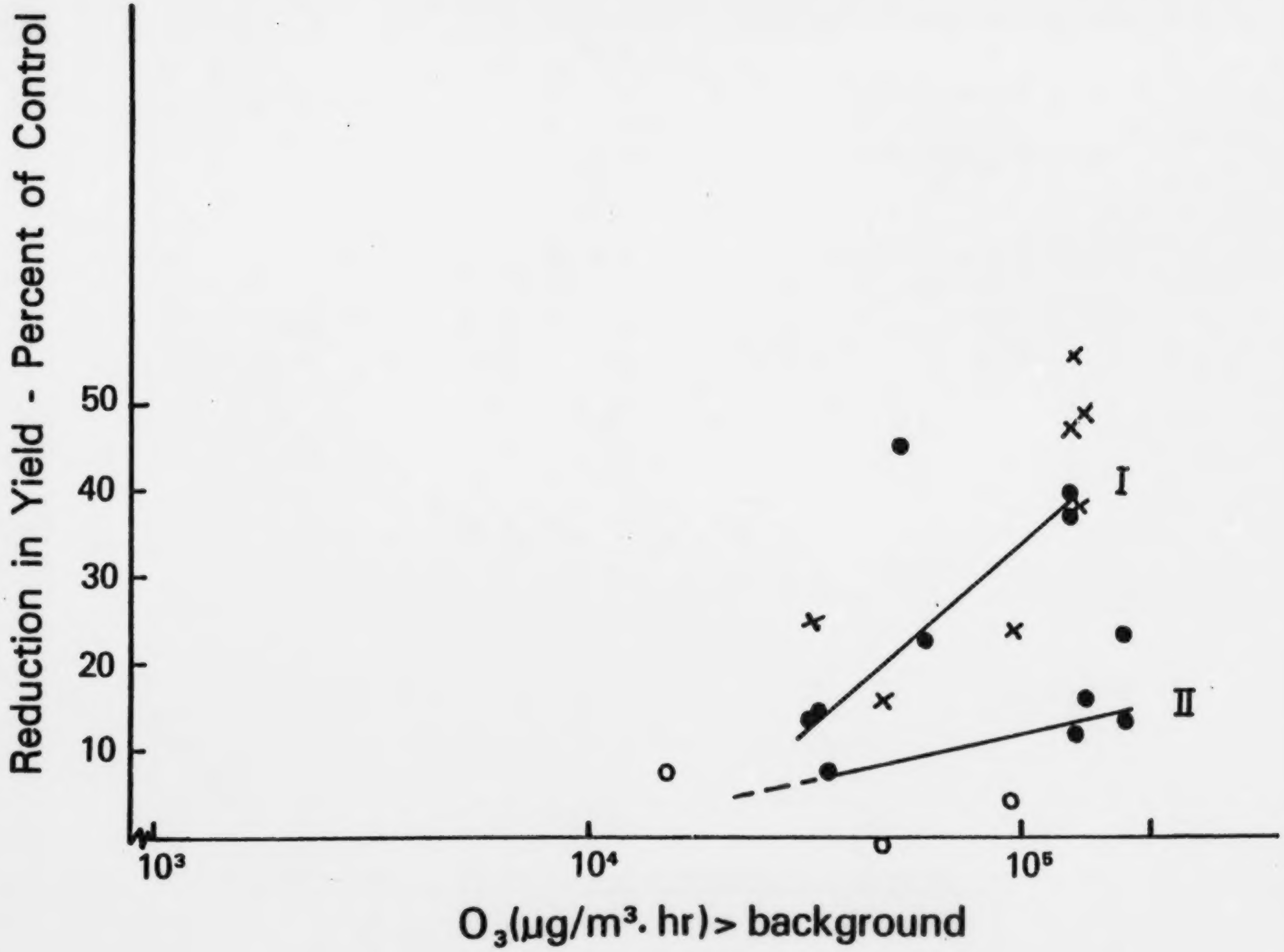

Figure V-4. Effects of chronic ozone exposure upon yield of corn. Data are from Oshima (1973), Heagle et al. (1972, 1979), and Thompson et al. (1976). Kernel dry weight yield is indicated by ●, stover dry weight by x. Non-statistically significant kernel weight data ($P<0.05$) are indicated by o. All stover data are statistically significant. Curve I connects kernel weight data from sensitive corn varieties (from Oshima 1973 and Heagle et al. 1979). Curve II connects kernel weight data from resistant varieties (from Heagle et al. 1979 and Thompson et al. 1976). Other points are from varieties of intermediate or very high sensitivity.

Losses in the yield of stovers, which are sold as fodder, generally exceed kernel yield loss in the varieties tested, but these data were excluded from the calculations. The assumption that only July and August exposures (62 days) affect yield is conservative with respect to the loss estimates, because cumulative exposures calculated from literature are based on 67 to 88 days of exposure.

Calculation of O_3 Loss Coefficient for Yield of Wheat

Heagle *et al*. (1979c) found that 0.05 ppm O_3 for 7 hours per day from shortly after emergence until maturity caused a 3% yield decrease in four varieties of winter wheat (Table V-3). A 16% yield loss occurred when the concentration was raised to 0.10 ppm. Bennett and Runeckles (1977) found that 0.09 ppm given 8 hours daily for 6 weeks caused 22% loss in dry weight of *Lolium multiflorum*, a member of the same grass tribe (Hordeae) as wheat. Converting the dryweight loss of *Lolium* to a yield loss in wheat from the regressions of Brisley and Jones (1952) suggests that a 19% yield loss may have resulted--very similar to the Heagle *et al*. (1979c) data.

A crop loss estimate for winter wheat must be based on ambient O_3 data for the late September-May period, which comprises much of the growing period of winter wheat in the Ohio River Basin (Illinois Crop Reporting Service 1977). During this period, growth of wheat is correlated with the accumulation of degree-days (above a certain minimum) principally during warm spells in October-November and April-May, often on days favorable for elevation of O_3 levels. The periods of active growth probably coincide with the period of pollution sensitivity.

Elevated O_3 levels occur for much of September and October in the Indianapolis area (Haering 1977), where the highest 1975 seasonal O_3 concentrations occurred in September at five of six monitoring stations, and in September and October in three of six stations in 1976 (Haering 1977). Elevated concentrations in April, May and October also occur in Illinois (Illinois EPA 1978). Data are lacking for the winter months, but if the cumulative O_3 exposure for April, May, and late September through mid November equaled three times the lowest of the three summer months in 1977, the cumulative exposure would be 60,700 $\mu g \cdot m^{-3}h$ over the 7 daylight hours of maximum sensitivity (Table V-2). This total equals an hourly mean of 0.048 ppm, similar to the mean concentration at which Heagle *et al*. (1972) found a 3% yield reduction (Table V-3). Although this yield reduction was found to be statistically non-significant, it compares favorably with the 4% weight reduction loss that can be calculated from the regressions of Brisley and Jones (1950) for *Lolium multiflorum* as a result of exposure to 0.03 ppm O_3 for 6 weeks (Bennett and Runeckles 1977). The cumulative exposure at New Albany (23,244 $\mu g \cdot m^{-3}h$ above background) exceeds the exposure used by Bennett and Runeckles (1977) and includes episodes surpassing 0.12 ppm O_3. In fact, the maximum hourly reading for 1975, 0.214 ppm, occurred on September 2, while the maximum 1976 concentration, 0.166 ppm, occurred on October 2 (Haering 1977). These concentrations may cause serious yield reductions according to

Shannon and Mulchi (1974), who found a 23 and 30% reduction in two wheat varieties in response to 0.2 ppm O_3 administered for just 4 hours per day for 1 week.

A yield loss coefficient of 6%, twice that estimated from Bennett and Runeckles (1977) data, reflects the higher O_3 exposures observed in the Midwest and is suggested as a most probable value. This estimate is obtained directly from the proportionality between the cumulative exposures used by Bennett and Runeckles (1977) and those estimated for New Albany. Since wheat is exposed to additional O_3 in November, and from March through May, additional damage may occur. A maximum loss function seems likely to be 10%. The 3% loss value of Heagle et al. (1979c), obtained under chamber conditions at 0.05 ppm O_3, represents a lower bound or minimum loss estimate.

O_3 LOSS FUNCTIONS RELATIVE TO AMBIENT AIR QUALITY STANDARDS

The O_3 loss functions developed in this section (summarized in Table V-6) imply that yield reductions occur at O_3 levels now permissible under EPA's revised secondary standard--0.12 ppm, expressed as a single annual peak hourly average (U.S. EPA 1979). Although our approach is consistent with EPA's conclusion that yield reduction from chronic O_3 exposure is of principal interest, several differences in assumptions lead to differences in conclusions:

(a) EPA used leaf-injury models to estimate yield reduction, an approach that neglects the increasing evidence of growth reduction occurring under chronic exposures without visible injury. EPA's assumption of a 5% visible injury threshold (before any growth reduction occurs) also is at variance with numerous research results cited in this section.

(b) The cumulative exposure technique used here incorporates frequent O_3 exposures which are high enough to affect plant growth, although not high enough to be associated with the "annual maximum peak concentration" used in the EPA interim O_3 review document (1979). The EPA approach assumes that the 6-to-8-hour average concentration accompanying the annual peak hourly average is a sufficient estimator of the extent of plant injury, thus ignoring chronic elevated O_3 levels common in rural areas of the eastern and western United States and exposures outside the 6-to-8-hour period. On this point, it should be noted that at New Albany, Indiana, as only one example, there were 24 days with O_3 hourly maximums between 0.08 and 0.10 ppm in the June through August period of 1977, and nine additional days with maximums between 0.10 and 0.12 ppm, levels clearly associated with crop yield effects. The new EPA standard operates on the implication that control within a 0.12 ppm annual hourly peak would prevent more than one 0.10 ppm 8-hour average from occurring each year, and a 0.10 annual hourly peak would prevent more than one 0.09 ppm 8-hour average from being exceeded. The new standard would not be violated during the nine days with maximums between 0.10 and 0.12 ppm, although these days could have 8-hour averages of 0.09 ppm (a level high enough to cause yield-related effects in some crop species), and produce a cumulative exposure of 8,550 $\mu g \cdot m^{-3}h$. The 24 additional days with an hourly maximum between 0.08 and 0.10 ppm could bring the cumulative exposure to 10,451 $\mu g \cdot m^{-3}h$, enough to cause crop effects in soybeans. Exposures on additional days would increase the O_3 exposure and the damage.

(c) EPA assumed that the pollution sensitivity of plants grown in the greenhouse exceeds that of field crops. As previously noted, this may hold for semi-arid regions, but not in the Ohio River Basin where precipitation and humidity are usually near optimum throughout the growing season.

(d) The EPA report also assumed that the most susceptible phenophase for non-foliar crops is during flowering and fruit development. Although this may be true, it does not mean that other phenophases are completely resistant and that growth effects will not occur during these stages which could affect yield (see Tingey et al. 1973b), especially in forage crops known to be sensitive.

CALCULATION OF ESTIMATED CROP LOSS COEFFICIENTS FROM SO_2 IN THE OHIO BASIN

Effects of SO_2 Exposure Upon Soybean Yield

Loss coefficients for SO_2 (rounded off to the nearest percent) would have to be related to the cumulative exposures associated with impact zones around the power plant point sources in the ORBES region. Impact Zone I (see Section VI) includes all of the area between the known or projected 0.05 ppm SO_2 isopleth and the 0.10 ppm SO_2 isopleth. Impact Zone I, therefore, is the area around the power plant where peak SO_2 concentrations are between 0.05 and 0.10 ppm at least twice each summer month. Impact Zone II similarly applies to the area exposed to 0.10 ppm SO_2 and above, twice per month. The method for determining impact zone locations and configuration is described in Section VI. The calculations assume that the same cumulative exposures occur in all zone I impact areas and slightly higher exposures in all zone II areas around present and projected Ohio River Basin power plants. Cumulative exposures were calculated for monitoring stations around the Clifty Creek power plant for these two zones, and from concentrations and exposure durations given in published reports of chronic yield effects studies. Data from studies that did not report effects on yield, or from which cumulative exposures could not be calculated, were not used in the analysis.

Only one study has been reported that describes the effects of chronic SO_2 exposure on soybean yield under experimental fumigation conditions in the field. The work was carried out in Kendall County, Illinois, 25 miles southwest of Chicago, in typical agricultural land (Miller et al. 1977, Sprugel et al. 1977, 1979). Plants were exposed for 2 years to controlled levels of SO_2 released from an open-air fumigation system. The SO_2 was released from pipes suspended 1 foot over the crop canopy and bled at controlled rates into the air current generated up-stream by compressors (Miller et al. 1977). Since other contaminants such as O_3 were probably present in the air during the experiments, it must be assumed that the plant responses represent an average integration of interactive effects (e.g. synergism, antagonism), if these are significant, of all the contaminants in the air. Thus, these yield losses can be interpreted as an effect resulting from the addition of SO_2 to air which contains other contaminants, rather than an effect resulting from the addition of SO_2 alone to clean air.

Concentration dynamics of the variable 4-to-5-hour fumigation episodes in the Kendall County work resemble those found during naturally occuring power plant fumigations over short averaging times (Miller et al. 1977, J. E. Miller, personal communication). Turbulence and fluctuating wind speeds led to an exposure regime with instantaneous variation in SO_2 about a mean, completely different from the uniform concentrations typically used in experimental studies. The concentration dynamics can be represented as a cumulative frequency that resembles the concentrations associated with power plant fumigations (Figure V-5).

Yield reduction in the Sprugel et al. work ranged from 6% at a mean exposure of 0.095 ppm in 1978, to 45% at a mean exposure of 0.79 ppm in 1977 (Table V-4). Chlorosis and bronzing were seen on plants exposed to 0.79 ppm, and mild chlorosis was seen on plants exposed to 0.36 ppm. No other treatments produced visible leaf injury. Thus, soybean yield was reduced regardless of whether visible leaf injury occurred.

In a study carried out near coal-fired generating plants in the Tennessee River Valley, Jones et al. (1979) made the following assumptions which could affect ability to recognize injury or damage to crops: (a) sufficient data exist for predicting the occurrence of visible SO_2 effects (e.g., leaf injury), but not effects without visible injury (e.g., physiological or biochemical changes); (b) growth and yield reduction occur only in plants exhibiting visible injury over more than 5% of the leaf area; and (c) an exposure threshold for visible injury exists. Thus, the study may be biased against detecting chronic reduction of growth and yield, which frequently occurs without any visible injury (Tingey et al. 1973b, Sprugel et al. 1979, Bell and Clough 1973).

According to the Jones et al. study, soybean yield may be either increased or decreased by exposure to 260 $\mu g \cdot m^{-3}$ (0.1 ppm) or more of SO_2, compared to plants exposed to less than 0.1 ppm SO_2. The study refers to Heagle et al. (1974), who reported that exposure to 0.1 ppm SO_2 did not cause statistically significant reduction of yield. However, the experimental design of Heagle et al. (1974) could not distinguish differences approaching 30% between treatment and control plots, apparently because of high plant-to-plant variability. Thus, losses of approximately 20% or less could have occurred, but were not detected in the statistical tests used by Heagle et al. The work of Sprugel (1977, 1979) and Miller (1977, 1979) on yield reduction shows that injury and yield loss effects can occur at concentrations interpreted by Jones et al. (1979) as satisfying control conditions.

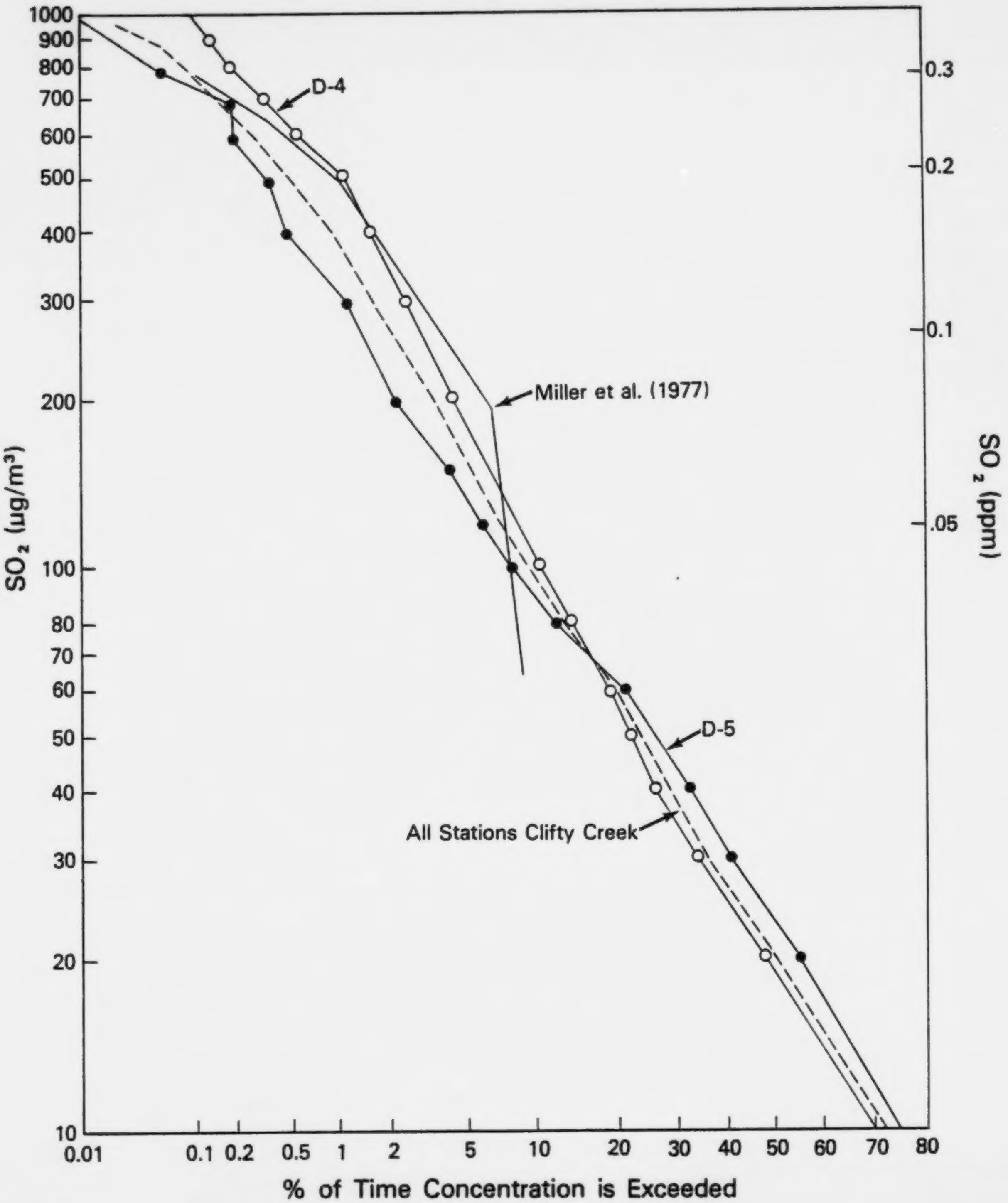

Figure V-5. Cumulative frequency curves for SO_2 monitoring stations inside Impact Zone I at Clifty Creek generating station (data from Teknekron Research, Inc.) and for the 0.-17 ppm exposure plot of Miller _et_ _al_. (1977).

TABLE V-4. EFFECTS OF CHRONIC SO_2 EXPOSURE UPON SOYBEAN YIELD. DATA ARE FROM CHRONIC EXPOSURE STUDIES USING CONCENTRATIONS WHICH ARE REACHED NEAR POWER GENERATING STATIONS IN THE OHIO RIVER BASIN.

	Days	Hours	Concentration (ppm)	Cumulative Exposure ($mg \cdot m^{-3}h$)	Effect
1.	Sprugel et al. (1977, 1979)				
	24	114(4.7)[a]	0.117	34.7	1977: 12.3% reduction in yield
	24	114(4.7)	0.300	88.9	20.5% reduction in yield
	24	114(4.7)	0.790	233.0	45.3% reduction in yield
	18	74(4.1)	0.095	18.4	1978: 6.4% reduction in yield
	18	74(4.1)	0.108	20.9	5.2% reduction in yield
	18	74(4.1)	0.192	37.2	12.2% reduction in yield
	18	74(4.1)	0.255	49.4	19.2% reduction in yield
	18	74(4.1)	0.362	70.2	15.9% reduction in yield
2.	Heagle et al. (1974)				
	43	258(6)	0.1	67.6	0% reduction in plant fresh weight compared to ambient air
	92	55(6)	0.1	143	10.9% decrease in pod fresh weight[b]; 1.9% increase in plant fresh weight[b]
	133	798(6)	0.1	209	17.8% decrease in seed harvest weight[b]; 12% reduction in pod fresh weight[b]; 5.8% increase in plant fresh weight
3.	Tingey et al. (1973b)				
	15	120(8)	0.05	15.7	7.6% increase in plant dry weight[b]
	15	120(8)	0.20	62.9	2.3% increase in plant dry weight[b]

[a] Number of hours/day of exposure

[b] Not statistically significant at P=0.05

The data reported by Jones et al. (1979) were not used in estimating loss functions for the ORB because the validity of their assumptions has not been clearly shown, and because there is a sharp dichotomy between their results and those of experiments free from their assumptions. Nevertheless, their report, and that of Noggle and Jones (1979), who showed that growth of cotton and fescue can be stimulated by addition of atmospheric sulfur, emphasize the importance of cultivation conditions in determining the response of plants to atmospheric pollutants.

If soil sulfur concentrations are low enough to limit growth, plants apparently can metabolize atmospheric sulfur, rather than accumulate it in toxic form. In conifers, a constant nitrogen:sulfur ratio in foliage can limit utilization of additions of either nutrient separately (Turner et al. 1979). Nitrogen fertilizer addition often results in sulfur utilization, but the desired growth response requires sufficient supply of sulfur. Thus, addition of large quantities of nitrogenous fertilizer can greatly increase sulfur demand and create a sulfur inadequacy, particularly in soils already low in sulfur. Sulfur deficiency has been observed in many southeastern states, including some in the Tennessee River Valley, according to a report by The Sulfur Institute (Coleman 1966). States of the Ohio River Basin are not included in Coleman's list, suggesting that soils within the Ohio River Basin are not generally characterized by sulfur deficiency. This conclusion is supported by an assessment of Indiana soils (Bertramson et al. 1950). However, local differences in soils and cropping practices could lead to localized soil sulfur insufficiency for certain crops.

The "fertilizing" effect of SO_2 gaseous uptake has been suggested in ryegrass pot experiments reported by Cowling et al. (1973). In a similar study of the effects of SO_2 additions upon ryegrass function, however, Bell and Clough (1973) noted that 0.067 ppm added daily for 6 weeks increased yield, but after 16 weeks the effect was reversed and significant growth reduction was observed. Thus, the sulfur fertilization effect may be reversed after plant requirements are met, even if an initial sulfur deficiency exists. This reversal of effects may explain the results of other experiments that have shown no deleterious effect or have shown growth increases in response to sulfur additions. For example, Tingey et al. (1973a) found that 0.05 ppm and 0.20 ppm SO_2 administered over 3 weeks caused no statistically significant effect upon soybean growth, although a small growth increase was suggested (Table V-4). Since the SO_2 treatment began 3 days after seeding and ended before blooming, effects on yield were not determined. If the experiment had been continued until the plants were mature, deleterious effects might have occurred once sulfur needs were met. Also, treatment of plants only during immature stages permits recovery through new leaf production, if plants are not exposed during sensitive flowering and fruiting stages (Guderian 1977).

Cowling et al. (1973) added 150 ppm of nitrogen to the soil in their experiments and repeated the application after each of three harvests.

This rate of addition is equivalent to applying nearly 400 kilograms/hectare ($kg \cdot ha^{-1}$) of nitrogen each time. In comparison, application rates of 45 to 90 $kg \cdot ha^{-1}$ are recommended for wheat, and 150 $kg \cdot ha^{-1}$ for corn in the U.S. (Chapman and Carter 1976). Thus, in the absence of evidence to the contrary, the sulfur defficiency induced under the experimental conditions of Cowling _et al_. (1973) would not be expected in the Ohio River Basin.

Heagle _et al_. (1974) showed that exposure to 0.1 ppm SO_2, together with O_3, caused a large decline in yield and growth of soybeans (50% to 70% reduction compared to controls), but 0.1 ppm SO_2 alone was much less damaging (Table V-4). The effects caused by SO_2 alone were not clear, however, because, as already described, losses as high as 27% were found to be statistically non-significant, apparently because of high plant-to-plant variability (Table V-4). The uncertainty in interpretation of SO_2 effects at lower concentrations reported in Heagle _et al_. (1974) made the results of limited value in predicting SO_2 effects in the Ohio River Basin. However, the losses caused by the presence of SO_2 and O_3 together indicate the SO_2 emissions in the vicinity of power plants could cause losses when in the presence of O_3 that would not occur in response to the SO_2 alone. Support for this conclusion was presented by Tingey _et al_. (1973b) who observed significant losses in freshweight of soybeans in the presence of mixtures of 0.05 ppm SO_2 and 0.05 ppm O_3, although exposures to the gases individually caused no significant growth effect.

Calculation of SO_2 Soybean Yield Loss Coefficients

The only data on soybean response at exposures comparable to those at coal-fired generating stations in the Ohio River Basin are those of the study in Illinois by Miller _et al_. (1977, 1979) and Sprugel _et al_. (1977, 1979). These results form much of the basis for the curves used for determining crop loss functions (Figure V-6). As already noted, these experiments were conducted in soybean fields where it is likely that a "background" of contaminants other than SO_2 also were present. Thus, the results should be interpreted as a response to the addition of SO_2 as it interacts with other contaminants already present in the air. The specific pollutant interactions cannot be determined because that was not the purpose of the study, and the composition of the air to which the SO_2 was added was not monitored.

Over the June-August 1977 period, cumulative SO_2 exposure near the Clifty Creek power plant reached around 110,000 $\mu g \cdot m^{-3}h$ at stations recording the highest exposures in Impact Zone II. Extrapolating from the yield reduction/exposure curve suggests that at least an 18% yield loss (the upper bound loss coefficient) could occur in soybeans growing in areas subject to these SO_2 exposures (second-highest hourly SO_2 concentrations during summer months averaging between 0.195 and 0.200 ppm). Much of this zone would be exposed to lower concentrations, however, since the largest area is just inside the 0.1 ppm isoleth. Here exposures would be one-half as high as exposures at the highest exposure stations. Thus, maximum losses here would approach 9%, if no sulfur deficiency exists (Figure V-6).

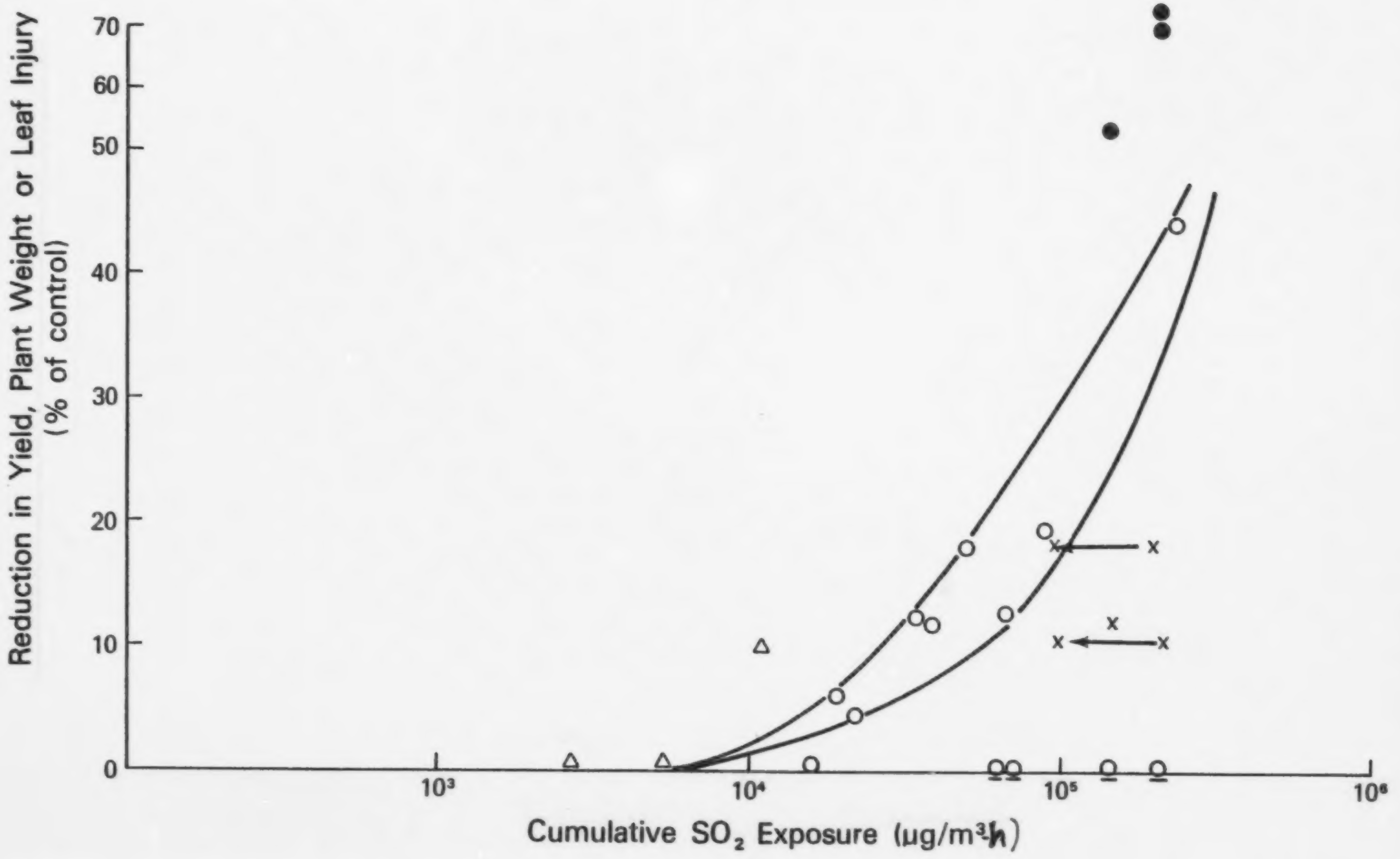

Figure V-6. Effect of chronic SO_2 exposure on soybean yield. Data from Sprugel et al. (1977, 1979) and Miller et al. (1977, 1979) (o); Heagle et al. (1974)(x=non-significant data, P=.05; ●= in presence of ozone); and Tingey et al. (1971, 1973) (Δ=leaf injury; o=plant weight). The two curves were fit to a minimum and maximum loss function for statistically significant yield data. The arrows connect the yield data of Heagle et al. (1974) for 133 and 92 days (reported) and for 62 days (calculated) of cumulative exposure. Further discussion is in the text.

If sensitivity to toxins occurs only during flowering and fruit development, as suggested by Heck and Brandt (1977), Guderian (1977), and Heagle *et al*. (1974), then the loss coefficients should be calculated only for exposure during July and August, the period of principal maturation and podding in the ORB. In Zone II of Clifty Creek, cumulative exposures average around 81,000 $\mu g \cdot m^{-3}h$ for July and August (Table V-2) at the high-exposure stations, and half that near the periphery of the zone, suggests yield losses would range from 14% down to 7%. Predicted damage from July and August exposures at Clifty Creek coincides well with studies by Sprugel *et al*. (1977, 1979) and Miller *et al*. (1977, 1979) who fumigated plants 24 and 21 times in July and August of 1977 and 1978, respectively. The plots exposed to a mean of 0.117 ppm in 1977 received a total dose during fumigations of 34,946 $\mu g \cdot m^{-3}h$. Exposures in the high-exposure stations at Clifty Creek during daytime fumigations reached 36,696 $\mu g \cdot m^{-3}h$.

The similarity of the exposure regimes also is seen in the cumulative frequency representation of exposures from Sprugel *et al*. (1977, 1979) and from Clifty Creek (Figure V-5). The yield losses in the 0.117 ppm plot were 12% (Sprugel *et al*. 1977), which serves as another estimate of yield loss at the high-exposure stations. Plants were exposed to 28,715 $\mu g \cdot m^{-3}h$, in excess of 0.1 ppm in the Sprugel and Miller study, compared to 17,859 $\mu g \cdot m^{-3}h$ at Clifty Creek. At Clifty Creek, however, an additional burden of 8,194 $\mu g \cdot m^{-3}h$ accrued from concentrations between 0.05 and 0.01 ppm, reaching a total of 26,054 $\mu g \cdot m^{-3}h$ cumulative exposure. Additional exposure to concentrations from 0.02 to 0.05 ppm SO_2 occurred on many days without trapping events at Clifty Creek, compared to a background of 0.005 to 0.015 ppm on non-fumigation days in the Illinois study (Sprugel *et al*. 1979). If the losses reported by Heagle *et al*. (1974) for 92 and 133 days of exposure were caused by exposure only during the 2-month (62 days) period of sensitivity, then the loss-exposure relationship for the Heagle *et at*. (1974) study and the Sprugel *et al*. (1977, 1979) study would be similar (Figure V-6).

The presence of somewhat elevated O_3 concentrations at one Kendall County, Illinois, site (Sprugel *et al*. 1977, 1979; Miller *et al*. 1977, 1979) suggests that the reported losses were due in part to interactions with O_3. Such damage would operate primarily during fumigation episodes, when control plants are also exposed to SO_2. Ozone reached 0.1 ppm only twice during the 1977 growing season and once during the 1978 growing season; concentrations between 0.06 and 0.10 ppm were reached six times during the experimental period (J. E. Miller, personal communication). Thus, O_3 levels are lower than in the ORB and would be expected to cause less damage through the study treatment sites. If the loss enhancement by O_3 is moderate, the loss response to 0.117 ppm SO_2 might be reduced from 12% to 9%. Thus the "most probable" loss function in the high-exposure areas of Zone II near ORB power plants can be estimated by taking the average of the corrected loss function from the Sprugel *et al*. (1977, 1979) study (9%) and the estimated loss from July-August exposures (14%), giving 12% as a conservative loss function. This figure is closely matched by the mean loss function derived from the two curves for the total daytime exposure during fumigations at Clifty Creek (36,696 $\mu g \cdot m^{-3} h$) (Figure V-6). For the periphery of Zone II, a 6% loss is probable.

If SO_2 only in excess of 0.1 ppm daytime exposure exerts a toxic effect on soybeans, cumulative exposure at the Clifty Creek high-exposure stations (17,859 $\mu g \cdot m^{-3} h$ for 8-hour days) would cause a yield loss of about 5% (the minimum loss coefficient), obtained as a mean from the two curves. Half that loss would be indicated for the outer edge of Zone II. The minimum loss estimate probably would be greater, however, if the curves were redrawn to depict the same losses, but only at exposures in excess of 0.1 ppm SO_2.

A single set of loss functions for Impact Zone II should reflect the change in SO_2 exposure with distance from the highest exposure zone and the greater area near the periphery. Thus the mean loss functions would be weighted towards the 0.1 ppm isopleth, suggesting that a conservative interpretation would place loss functions at 3%, 9% and 6% for Zone II (see Table V-6).

Loss estimates in Zone II were reduced proportionately to reflect the lower exposures in Zone I, giving 1%, 4% and 3% for the minimum, maximum and most probable values. A proportionate reduction implies that the exposure-loss relationship found at higher exposures holds at lower exposures. Although this assumption is untested, no field data support the concept of a "threshold" for chronic exposure to SO_2 at 0.1 and 0.05 ppm. Jones _et al_. (1979) suggest that a threshold could exist under conditions of sulfur limitation. Even though plants may metabolize or detoxify SO_2 at lower concentrations, the work of Black and Black (1979) and Unsworth _et al_. (1972) shows this may not occur without physiological cost. The predicted loss coefficients reflect this uncertainty in the minimum and maximum limits, with the most probable value falling between the two extremes (see Table V-6).

Effects of SO_2 Exposure Upon Wheat Yield

Field experiments on the chronic effects of SO_2 exposure on wheat, under environmental conditions similar to those of the Ohio River Basin, could not be found in the literature. However, observations have been made of wheat growing in the vicinity of pollutant sources in Germany. These studies involved observations during 7-month growing seasons in 1959 and 1960 (Guderian 1977, Heck and Brandt 1977). Average SO_2 concentrations of 0.129 ppm in 1959 and 0.145 ppm in 1960, associated with maximum 30-minute averages of 5.4 and 6.6 ppm, respectively, caused yield reduction in wheat (reported in U.S. EPA 1973). Guderian (1977), drawing on results of the same study, projected a threshold concentration of 0.24 to 0.28 ppm for effects on growth, yield, or quality of winter wheat during exposure times, and 0.009 to 0.024 during his entire "monitoring time." These threshold concentrations for winter wheat are lower than for species such as alfalfa, clover, oats, and summer wheat. Since cumulative exposures and amount of yield reduction are not reported in available publications, the results of the German study could not be used in determining crop loss functions for the ORB.

Brisley and Jones (1950) fumigated field-grown wheat in Arizona with SO_2 from 1941 through 1944. Yield reduction correlated linearly with percent of destroyed leaf area, but the SO_2 concentrations causing the reductions were not reported. Data suggest that variation in growing conditions over the 4 years of the experiments (mean yield per plot ranged from 34 to 55 ounces) would have required large sample sizes and accurate measurement of SO_2 concentrations to demonstrate exposure effects.

Under laboratory conditions, and using potted plants, Laurence (1979) found that seven cultivars of wheat showed leaf injury in response to a range of concentrations from 0.2 to 0.6 ppm SO_2, administered continuously over 30 to 100 hours. No correlation was found between leaf injury and growth reduction. Growth increase was found at 0.2 ppm, and no significant growth reduction--compared to controls--was observed at any of the exposures. However, statistically significant growth reduction occurred in plants exposed to 0.4 and 0.6 ppm SO_2 for 78 and 100 hours. This work raises questions about the importance of available sulfur in the soil and whether the apparent "S fertilization" is a function of a large sulfur demand brought on by rapid growth stimulated with fertilizer addition and optimum growth conditions. Because Laurence's work was carried out over only several days, it is uncertain whether the sulfur enhancement effect would continue. Bell and Clough (1973) found sulfur growth enhancement in ryegrass (*Lolium perenne*) for 6 weeks of exposure, but after 19 weeks, significant growth retardation was observed. The several-day exposure period of Laurence (1979) is long enough to characterize acute effects or effects from one or several episodes, but cannot be considered a long-term exposure (exposure period as long as the life cycle or critical phenophase). Because Laurence worked with seedlings, neither chronic nor acute exposure effects on yield could be determined. The same exposures during flowering or seed production might have caused reduction if the relationship between yield and amount of destroyed leaf area found by Brisley and Jones (1950) and Bell and Clough (1973) is accurate.

In the absence of yield effects data for wheat, results from a series of experiments conducted with ryegrass (*Lolium perenne* and *L. multiflorum*) and orchard grass (*Dactylis glomerata*), related to wheat, were examined for use in estimating potential wheat yield reduction in the ORB. Experiments which included clipping (periodic harvest) were not used because clipping may offset the inhibitory effects of SO_2 exposure (Bell *et al.* 1979) and because the practice does not generally occur in wheat cultivation. The series of experiments (Table V-5) involved treatment of plants with chronic (4 to 46 weeks) doses of low SO_2 concentrations (.023 to 0.12 ppm) similar to those sometimes found near power plants in the Ohio River Valley (Table V-5). Continuous exposures to a uniform SO_2 concentration were used. These data are valuable because of the low concentrations and long exposure times used. Injurious effects show a linear relationship to cumulative exposures.

TABLE V-5. EFFECTS OF CHRONIC EXPOSURE TO SO_2 UPON DRY WEIGHT OF *Lolium perenne* L., AND *Triticum sativum*. DATA ARE FROM CHRONIC EXPOSURE STUDIES USING CONCENTRATIONS WHICH ARE REACHED NEAR POWER GENERATING STATIONS IN THE OHIO RIVER BASIN. ONLY EXPERIMENTS WITH CUMULATIVE EXPOSURES OF 200,000 $\mu gm^{-3}h$ OR LESS ARE GIVEN.

Weeks	Hours	Concentration (ppm)	Cumulative Exposure ($\mu g \cdot m^{-3}h \times 10^3$)	Effect
1. Ashenden & Mansfield (1977) - *Lolium perenne* (ryegrass)				
4	672	0.11	194	17.6% and 33.8% shoot loss in two experiments at different temperatures; windspeed greatly influenced plant response.
4	672	0.11	194	
2. Crittenden & Read (1978) - *Lolium perenne*				
8	1344	0.027 (0.053)[a]	95	36% reduction in shoot dry-weight (sand culture)
19	3192	0.023 (0.053)	192	20% reduction in shoot dry-weight (sand culture)
12	2016	0.026 (0.099)	139	24.6% reduction in shoot dry-weight (unfertilized sandy loam
17	2856	0.023 (0.053)	180	25.6% reduction in shoot dry-weight (fertilized sandy loam)
3. Bleasdale (1952) - *Lolium perenne*				
9	1512	0.047 (0.1-0.2)	186	47.0% reduction in shoot dry-weight (mean of two fertilizer levels)
4. Bell *et al*. (1979) - *Lolium perenne*				
6.3	1056	0.047	129	25% reduction in shoot dry-weight in Hurley soil
6.3	1056	0.047	129	20.6% reduction in shoot dry-weight in Afton soil
5. Laurence (1979) - *Triticum sativum* (wheat)				
	30-100	1.2-0.6	15.7-157	Cumulative exposure to affect one-half of population of two winter-wheat cultivars = 78.6-83.8 x 10^3 $\mu g \cdot m^{-3}h$

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[a] Occasional peak values; for some weeks, mean values were below that shown

[b] Calculated, weighted mean from author's data

However, the intermittent fumigations characteristic of power plants expose vegetation to a dynamic SO_2 regime. Continuous exposure regimes lack both the periods with only background SO_2 and the fluctuating concentrations found during fumigations. Intermittent exposures may permit partial or complete recovery from injury, depending on the relative length of time between pollutant-free and exposure periods (Guderian 1977), but the relationship has been worked out best only for exposure over several hours. For a given concentration, recovery may not occur or may be retarded if a definite exposure duration has been exceeded or if the pollutant-free period is too short. These findings cannot be related directly to chronic exposures, but the continuous hourly or daily exposure periods of the ryegrass and orchard grass experiments may overestimate damage to wheat in the Ohio River Valley.

On the other hand, peak concentrations, which occasionally reach 0.2 to 0.5 around ORB power plant sites, may cause greater damage than the same total amount of SO_2 in lower concentrations over longer time periods (Guderian 1977, Heck and Brandt 1977). Thus, continuous exposure experiments without SO_2 peaks may imply less damage than actually exists in the vicinity of pollutant sources. Clearly, then, most experiments reported in the literature do not mimic the exposure regime characteristic of field conditions, and thus may introduce error into loss predictions. The net magnitude of the error cannot be assessed, but by encompassing the range of results into yield loss estimation as a function of cumulative exposure, some, if not all, of the disparity can be reduced.

Differences in cultivar sensitivity may be important also in determining losses in wheat and related range and pasture grasses. Bell and Clough (1973) reported losses of over 40% in dry weight in *Lolium perenne* S23, a cultivated variety, but continuation of the work showed no effect upon the indigenous variety Helmshore (Bell and Mudd 1976). In striking contrast, Crittenden and Read (1978) compared both varieties in a series of SO_2 exposure experiments and found Helmshore to be no more resistant.

Two varieties of winter wheat showed greater leaf injury in response to 0.2 to 0.6 ppm SO_2 over 30 to 100 hours than did five varieties of summer wheat, according to Laurence (1979), but Guderian (1977) reports that the threshold for loss of growth and yield for winter wheat is higher than for spring wheat. These inconsistencies, plus the recent work by Ashenden and Mansfield (1977) showing the significance of boundary layer resistance, emphasize the importance of cultural and environmental conditions in determining biological responses to pollutant stress, and emphasize the value of presenting yield predictions as a range of values.

Calculation of SO_2 Yield Loss Coefficients for Wheat

Loss functions for wheat were based principally on the response of *Lolium perenne*, disregarding the more sensitive *Dactylis glomerata* L. The taxonomic affinity of the genera *Lolium* and *Triticum* is the basis for assuming that sensitivity to SO_2 would be similar in the three genera. This assumption has never been tested, but is supported by EPA classification (1973) of the two species in the sensitive susceptibility group, rather than the intermediate or resistant groups.

To convert loss of dry weight in ryegrass and orchard grass to reduction in wheat, conversion factors were adopted from Brisley and Jones (1950), who established a linear regression relationship between yield loss and percent of foliage area destroyed for wheat growing in Arizona. Yield loss was only slightly less sensitive to SO_2 exposure than was amount of leaf area destroyed. Bell and Clough (1973) also report close correspondence in the response of leaf area and leaf dry weight in ryegrass: leaf area declined 44% and 51% in two experiments, compared to 46% and 57% declines in leaf dry weight. Thus, yield loss can be correlated with dry weight reduction.

In the Ohio River Valley, the wheat grown is almost exclusively winter wheat. In Illinois, the ninth largest wheat-producing state, most winter wheat is seeded in early October, produces fruiting heads in May, and is harvested in late June and early July (Illinois Cooperative Crop Reporting Service 1977). Thus, wheat plants could be exposed to SO_2 from October through June. Winter exposure does not confer much resistance, according to Bell _et al_. (1979), who found losses similar in winter and summer. Winter sensitivity probably occurs because winter wheat and other winter annuals can assimulate CO_2 and, presumably, SO_2 at temperatures below the freezing point (Guderian 1977).

The cumulative exposure for winter wheat in the vicinity of power plants in the Ohio River Valley was calculated from the estimated SO_2 loads for the months of October, November, and March through June (Table V-2). The period from December through February was not included because snow cover was assumed to completely eliminate exposure to SO_2, (although this is often not the case in the ORB). The estimated cumulative exposure of 104,000 $\mu g \cdot m^{-3}h$ SO_2 at Clifty Creek (Zone II) is close to the range of cumulative exposure in the reported experiments (Tables V-2 and 5). The mean hourly exposure at Clifty Creek during the winter wheat season is estimated at about 0.009 ppm, about one-third the concentration used by Bleasdale (1952) and Crittenden and Read (1978). Here, as with soybeans, the same questions apply about the importance of continuous vs. intermittent exposures and the amount of additional damage to be expected from peak exposures. Few data exist to answer these questions for low-level chronic exposure conditions.

The experiments on _Lolium_ spp. and other grasses demonstrate that losses of shoot dry weight often reach 30-50% (Crittenden and Read 1978, 1979; Bell _et al_. 1979), but the cumulative exposures used (up to 10^6 $\mu g \cdot m^{-3}h$) frequently far exceeded those near power plants. Cumulative exposures of 200,000 $\mu g \cdot m^{-3}h$ or less, however, show that substantial shoot weight reduction in ryegrass is to be expected (Table V-5). Annual ryegrass and orchard grass are even more sensitive (Crittenden and Read 1978). The data from perennial ryegrass alone suggest that the exposures in the high-exposure area of Clifty Creek Impact Zone II could cause a shoot weight reduction from 17.6 to 36%, assuming the very high value from the early work by Bleasdale (1952) to be an outlier. The estimated yield losses calculated from Brisley and Jones (1950) for the reported shoot weight reductions are 10-20% and 7-12%, for two different regression equations. The intermittent exposure regimes around power plants may permit partial growth recovery, suggesting that 7% and 12% serve as the minimum and maximum loss functions for the high-exposure area, and 3% and 6% for the entire zone II. A mean of 10% is used as the most probable loss function for the high-exposure area, with 5% a conservative loss function applied to the entire Zone II (Table V-6). As with soybeans, the loss functions are interpreted conservatively to mean that the loss function estimates the loss due to the addition of SO_2 to air contaminants with O_3 and other pollutants present.

The basis for estimating a loss function in Zone I is less certain than for Zone II. The following choices could be argued regarding plant response in lower exposure areas: (1) No effects occur because there are no data on yield and growth at the cumulative exposures to be expected, or because an unproven damage threshold is higher than most exposures, or (2) deleterious effects occur with damage declining with exposure. Alternative (1) is rejected because the effects data show substantial losses at these and moderately higher concentrations (Ashenden 1979, Ashenden and Mansfield 1978, Crittenden and Read, 1978, 1979, Bleasdale 1952), and because there is no evidence of a regional yield reduction threshold. Alternative (2) is accepted, assuming no soil sulfur deficiency, and assuming small yield losses begin to occur when SO_2 levels rise substantially above background. In the absence of conflicting information, loss functions are proposed in proportion to the cumulative exposures, i.e., to 1% for minimum, 3% for the maximum, and 2% for the most probable loss. These values reflect the damaging levels of NO_2 and O_3 (Teknekron, Inc., unpublished data) and synergistic effects of NO_2 in the presence of SO_2 at concentrations found in Zone I (Ashenden 1979, Ashenden and Mansfield 1978). Synergisms plus extreme air pollutant sensitivity in wheat during anthesis (Shannon and Mulchi 1974) may occur simultaneously, creating maximum susceptible conditions, although in some years these conditions might not occur.

Calculation of SO_2 Yield Loss Coefficients for Corn

Corn (*Zea maize*) is reported to be one of the most resistant crop plants among those studied by Laurence (1979) and the U.S. EPA (1973). Laurence exposed corn seedlings to SO_2 concentrations ranging from 0.1 to 0.5 ppm over 6 to 100 hours in growth chambers under controlled conditions. Although data were not presented, Laurence reported that leaf injury was "minimal" at all exposures, and leaf and stem weight showed no statistically significant reduction in exposed plants. The EPA (1973) classified corn among resistant plants, based on observations at Sudbury, Ontario. The resistant category contains plants for which the threshold for plant injury exceeds 2 ppm for 8 hours exposure, or 8 ppm for a 1-hour exposure.

Data from the short-term studies of Laurence (1979) and the studies cited by U.S. EPA (1973) suggest that damage to corn should be minimal in the vicinity of Ohio River Valley power plants, but the possibility of additional effects on yield from chronic SO_2 exposure in the presence of elevated O_3 cannot be discounted. Unsworth et al. (1972) found that SO_2 concentrations as low as 0.01 ppm for 10 minutes reduced stomatal resistance in corn in the light or in the dark, increasing plant water loss. During dry years, a growth stress might be expected, particularly since increased stomatal resistance continued for 24 hours. No other data describing corn were located, but Black and Black (1979) showed that decreases of stomatal resistance in broad bean, induced by low SO_2 concentrations, are accompanied by greater death of leaf epidermal cells than in untreated plants. Black and Unsworth (1979) demonstrated that decreased stomatal resistance is accompanied by increased water loss and lower rates of photosynthesis than in untreated plants. From June through August, the mean hourly SO_2 concentration at Clifty Creek power station is 0.018 ppm--nearly twice the level Unsworth et al. (1972) found to cause increased stomatal opening. Although some results indicated that, in certain species, effects of SO_2 on stomatal resistance are dependent on atmospheric humidity (Mansfield and Majernik 1970), further data are needed for field conditions.

The liklihood of decreased stomatal resistance, accompanied by the other responses described, must be considered high for corn during at least part of the growing season. Damage also may be indicated at concentrations greater than 0.2 ppm, since the U.S. EPA (1973) classified corn as a resistant species, requiring 0.2 to 2.5 ppm for 8 hours to reach a threshold for injury. At Clifty Creek, about 9 hours of exposure occur in excess of 0.2 ppm each year in the high-exposure area. These limited data indicate that injury to corn from the addition of SO_2 is probably smaller than to wheat or soybeans, but could reach 5% in the high-exposure area of Zone II, averaging 3% for the entire zone. The possibility that O_3 further enhances damage caused by SO_2, as well as loss from chronic decrease in stomatal resistance, cannot be ruled out and should be investigated. Suggested losses reflect the effect of changeable climatic and cultivation conditions which would cause year-to-year variation in loss functions and a lack of any basis for the existence of threshold pollutant concentrations (Table V-6).

CONCLUSIONS

O_3 and SO_2 pollution in the Ohio River Basin is being observed at levels which can cause important reductions in crop yields (Table V-6). Analysis of existing data on pollutant effects suggests that elevated O_3 concentrations over the region are causing widespread yield reductions in soybeans, corn, and wheat, in comparison to potential clean air yields for the same agricultural management practices. Field losses ranging from 6% in wheat to 14% in soybeans are reported as probable responses to O_3 exposures during the growing season.

TABLE V-6. ESTIMATED CROP LOSS COEFFICIENTS FOR CHRONIC SO_2 AND O_3 EXPOSURE. THE BASIS FOR EACH OF THE LOSS COEFFICIENTS IS GIVEN IN THE TEXT.

Crop	Impact Zone (ppm SO_2)	Loss Coefficients (% yield reduction) Minimum	Maximum	Probable
SO_2				
Soybeans	II (0.1+)	3	9	6
	I (0.05-0.1)	1	4	3
Corn	II (0.1+)	0	3	2
	I (0.05-0.1)	0	2	0
Wheat	II (0.1+)	3	6	5
	I (0.05-0.1)	1	3	2
OZONE				
Soybeans		5	26	14
Corn		4	15	8
Wheat		3	10	6

Sulfur dioxide emissions from coal-fired electric generating stations can cause local, but still significant, crop effects. In the vicinity of power plants, the aggregate effect of exposure to SO_2 and O_3 can cause yield reductions in corn and wheat exceeding 10% or even 15% compared to a clean air yield. Yield losses in soybeans may reach 20% near power plants.

Analysis of existing data suggests that yield reductions over the region are caused principally by O_3 exposure. Direct validation of the loss function model presented here can only be achieved through long-term, intensive pollutant effects studies in Ohio River Basin croplands. Although this site-specific research has not been done, partial validation for the model can be found in crop studies conducted elsewhere in the eastern U.S. where O_3 loading and crop growing conditions are similar to the Ohio River Basin. Research results which support the findings of O_3-induced yield losses presented in this chapter come from studies of soybeans, corn, potatoes, and tomatoes in Maryland and Virginia (Heggestad et al. 1977, Heggestad 1980), soybeans in North Carolina (Heagle et al. 1979c), soybeans in Minnesota (Kohut et al. 1979), and beans and tomatoes in New York (MacLean and Schneider 1976). Yield losses similar or greater to losses estimated for the Ohio River Basin have been reported in these studies to be caused by ambient O_3 exposures similar to, or lower than those in the Ohio River Basin.

Although some interactions of multiple pollutant exposures undoubtedly have been accounted for in this analysis by using data from experiments conducted with air containing field levels of contaminants for part or all of the experimental period, the importance of interactive effects has not been critically determined. A much fuller knowledge is needed of the potential for pollutant interactions to affect crop yield, particularly near point or urban pollutant sources. Further data also are needed to determine whether the additive and synergistic effects observed in controlled environments occur in the field under the conditions of reduced boundary layer thickness.

Because of uncertainties in the existing data, and variations in varietal and phenophase sensitivity, the minimum crop loss estimates are considered to be low estimates of probable lower-bound effects because they ignore the other factors influencing susceptibility. The upper-bound estimates reflect possible plant response during sensitive life-cycle stages and growing conditions in conjunction with deleterious meteorological conditions. These coinciding conditions may not occur every year, but when they do, larger yield reductions should be expected.

Statistical analyses of the literature data might refine these loss estimates, but testing the assumptions underlying the analysis will require an improvement in the data base. Studies must be conducted in the field, with realistic pollutant exposure regimes and concentrations, statistically rigorous experimental designs, and season-long exposures.

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SECTION VI

CALCULATION OF AREAS OF DIRECT SULFUR DIOXIDE IMPACT AROUND COAL-FIRED POWER PLANTS

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INTRODUCTION

Assessing potential environmental impacts of a pollution source requires identification of the area in which pollutant concentrations are sufficiently high to produce adverse effects. The area subject to vegetation damage by sulfur dioxide (SO_2) emissions from coal-fired power plants should be defined as the area in which ambient SO_2 concentrations are high enough to affect the yield of sensitive species. To determine such an impact area requires knowledge both of actual SO_2 concentrations around the power plant and of the relationship between SO_2 concentrations and vegetation responses. The impact zone should include the area where the actual SO_2 concentration exceeds the level associated with damage or yield reduction.

The effects of SO_2 on vegetation are not completely known, but the literature synthesis (Section III) and dose/response functions (Section V) show that periodic short-term exposures at concentrations as low as 0.05 ppm SO_2 may cause growth reduction of sensitive species such as alfalfa, soybeans, and white pines. Consequently, 0.05 ppm was chosen as the lowest concentration of interest for this study. The other information required for area impact evaluation, ambient SO_2 concentrations around a source, can be expressed in the form of concentration isopleths, with the boundary of the largest potential impact zone defined by the 0.05 ppm isopleth.

Estimation of SO_2 concentration isopleths around a point source, such as a coal-fired power plant, is usually done by applying plume dispersion models: mathematical expressions of the atmospheric transport and dispersion of air pollutants as a cone expanding with distance (Turner 1979). Inputs to a dispersion model include meteorological and emissions data; outputs consist of estimated pollutant concentrations at various distances from the source. Since a large number of data points can be generated by a dispersion model, preparation of concentration isopleths is relatively easy.

A major disadvantage of dispersion models has been the unreliability of model predictions. Almost all existing dispersion models are based upon the assumption that the plume spread follows a Gaussian distribution

(Turner 1979). A Gaussian plume is a reasonable approximation when there is a uniformly varying temperature with elevation, but it is less reliable when there are discontinuities in the stability of atmospheric layers, also referred to as atmospheric inversions. Under these conditions, pollutants can be trapped below the inversion layer, and ground level SO_2 concentrations can be very high (Carpenter et al. 1971). Inversion events are significant because the resulting high surface SO_2 concentrations may cause crop yield loss.

Another factor affecting the accuracy of the Gaussian plume model is wind speed. Ragland et al. (1979) compared model predictions to measured SO_2 concentrations around the Columbia power plant in Wisconsin and found that the model tended to underestimate SO_2 when winds were light, and the integrity of the plume was disrupted by isolated wind puffs. In general, the Wisconsin study found the correlation coefficient between monitored and predicted hourly concentrations to be 0.36. Given this inability of existing dispersion models to predict accurately the high SO_2 concentrations which produce effects on vegetation, an alternative method is desirable for estimating SO_2 concentration during pollutant trapping incidents.

Another method for estimating SO_2 concentration isopleths is ambient air quality surveillance. This requires a well-designed network of SO_2 monitoring stations to measure actual ambient SO_2 concentrations, which are then used to establish isopleths around the pollution source. Because air quality data reflect both actual emissions and meteorological conditions, the advantage of using them is that the actual pollution situation is being described. This method is particularly useful in the case of atmospheric inversions; maximum hourly SO_2 concentrations from each monitoring station can be used to construct concentration isopleths for these extreme conditions.

The purpose of the work reported in this section is to develop a model, based on measured SO_2 concentrations around five selected power plants, which can be used to predict SO_2 concentrations around other present and future power plants in the ORBES region. The basic procedure involves (1) determining the general shape of concentration isopleths at representative power plants having an SO_2 monitoring network and finding quantitative relationships by which to calculate the area of the isopleths of interest, e.g., 0.05 ppm, and (2) developing a method of extrapolation to find the impact area of present and future power plants having no SO_2 monitoring network.

SO_2 CONCENTRATION ISOPLETHS AROUND POWER PLANTS WITH MONITORING NETWORKS

There are 10 power plants or groups of power plants in the ORBES region for which American Electric Power (AEP) operates air-quality monitoring networks (Table VI-1). Each plant is unique in its emission rate, surrounding topography, and presence of nearby sources of SO_2. The monitors are located so that those to the northeast of the plant tend to monitor the impact of the plant, those to the southwest tend to monitor

TABLE VI-1. AEP POWER PLANTS WITH SO_2 MONITORING NETWORKS (Teknekron 1979)

Power Plant(s)	Average Yearly SO_2 Emission Rate (g/s)	
Muskingum	9,486	
Big Sandy	1,657	
Kammer - Mitchell	10,107	
Amos	3,375	
Gavin - Kyger - Sporn	18,467	
Philo	723	(1975 only)
Breed	2,165	
Tanner's Creek	4,316	
Clifty Creek	8,622	
Cardinal - Tidd	4,143	

background pollutant levels upwind of the plant. The stations to the southeast tend to measure the impact of the plant during persistent northwesterly winds, generally in the winter (Marek 1976).

Five power plants were used to develop an algorithm to calculate the SO_2 distribution: Big Sandy, Breed, Clifty Creek, Muskingum River, and Gavin-Kyger Creek-Sporn complex (Figures VI-1 to 5). The other five power plants were excluded for a variety of reasons. At Cardinal-Tidd and Kammer-Mitchell, for example, the topography was very irregular, and the monitors were located on hilltops where crops are unlikely to be found. In addition, many of the highest SO_2 concentrations appeared to be direct plume strikes on the monitor rather than more general fumigations. Sulfur dioxide from other sources was a problem at Kammer-Mitchell, Amos, and Tanner's Creek. The Philo power plant ceased operation in 1976 during the monitoring period.

Hourly SO_2 concentrations at each monitor around the five plants were supplied by Teknekron, Inc. An assumption was made that the summer concentrations (June, July, August) were most important for crop effects and that elevated SO_2 concentrations at least twice per month might be required to induce plant damage (see Sections III and V). Therefore, the mean second-highest concentration at each monitor during June, July and August of 1975-1977 (nine values per monitor site) were plotted on maps of the monitoring network. Using these plotted values and the patterns evident between networks, preliminary isopleths of SO_2 concentration could be drawn freehand for the area in and adjacent to the monitoring network (Figures VI-6 to 10).

The method of presenting SO_2 concentrations by averaging second-highest values can be compared to other ways of summarizing SO_2 data. In Figure VI-11, the mean second-highest monthly concentrations are compared to the mean highest SO_2 concentrations every two weeks (18 data points) at each monitor around six different power plants. Figure VI-12 shows the comparison with the mean highest three-hour average each month (9 data points each monitor). Considering the diversity of monitoring situations (emissions, topography, etc.), the correspondence is quite good. This indicates that any of the three methods could be used to summarize the data. More importantly, the method chosen is biologically meaningful in that recovery time for some plants following a high SO_2 exposure is thought to be about two weeks. Moderate SO_2 concentrations occur frequently throughout the summer at these stations, and exposures are of sufficient duration (one to several hours) to cause damage.

The SO_2 concentrations during fumigation events regularly show an elliptical distribution skewed northeasterly from the power plant. This is thought to be due to the meteorology during the stagnation events, in which the pollutants are trapped in the lower atmosphere in air masses moving slowly from the southwest (Teknekron 1979). Since crops may be damaged by concentrations of SO_2 lower than those being observed (as low as 0.05 ppm), it was necessary to model pollutant distribution beyond

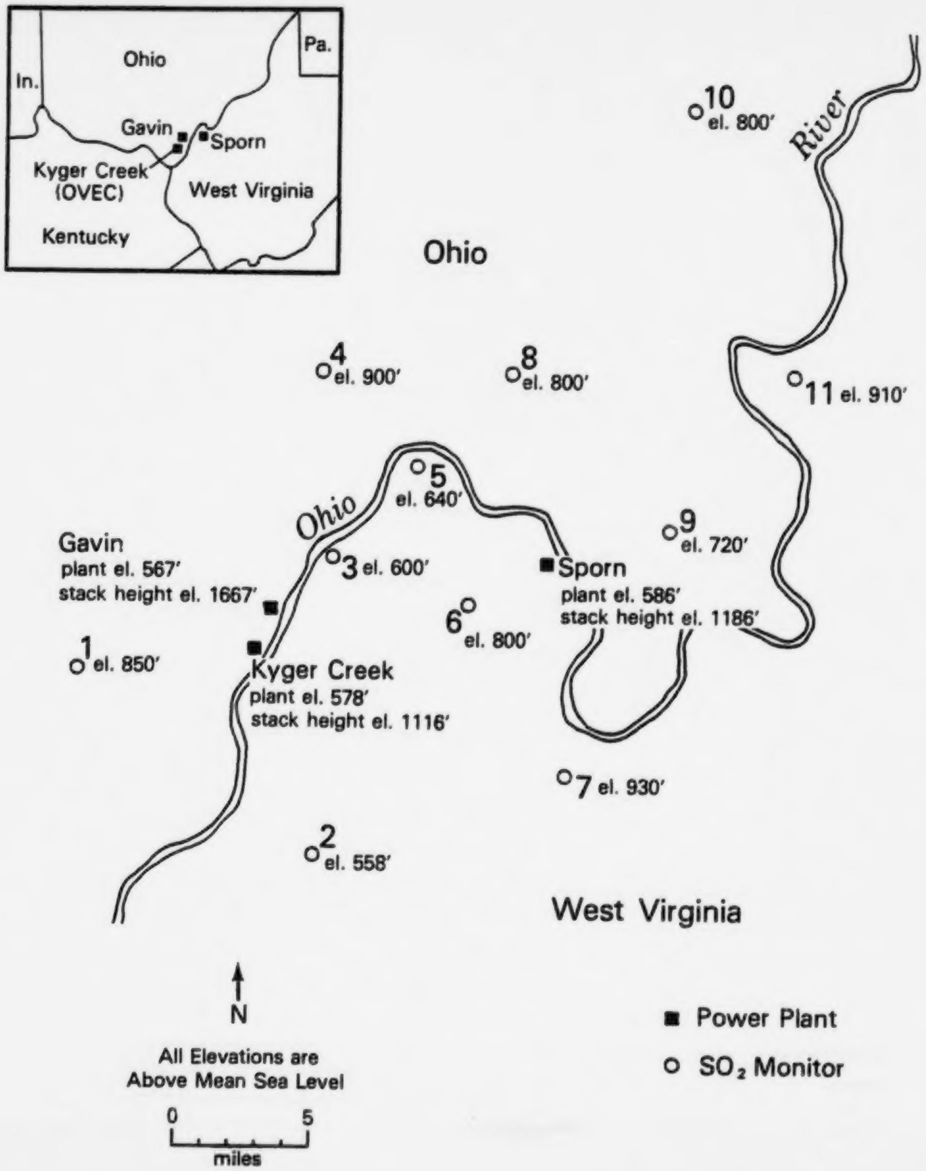
Ohio
Pa.
In.
Gavin
Sporn
Kyger Creek
(OVEC)
West Virginia
Kentucky
Ohio
River
10 el. 800'
4 el. 900'
8 el. 800'
11 el. 910'
5 el. 640'
Ohio
9 el. 720'
Gavin
plant el. 567'
stack height el. 1667'
3 el. 600'
Sporn
plant el. 586'
stack height el. 1186'
6 el. 800'
1 el. 850'
Kyger Creek
plant el. 578'
stack height el. 1116'
7 el. 930'
2 el. 558'
West Virginia
N
All Elevations are
Above Mean Sea Level
0
5
miles
■ Power Plant
○ SO₂ Monitor

Figure VI-1. AEP monitoring network around the Gavin-Kyger Creek-Sporn power plants.

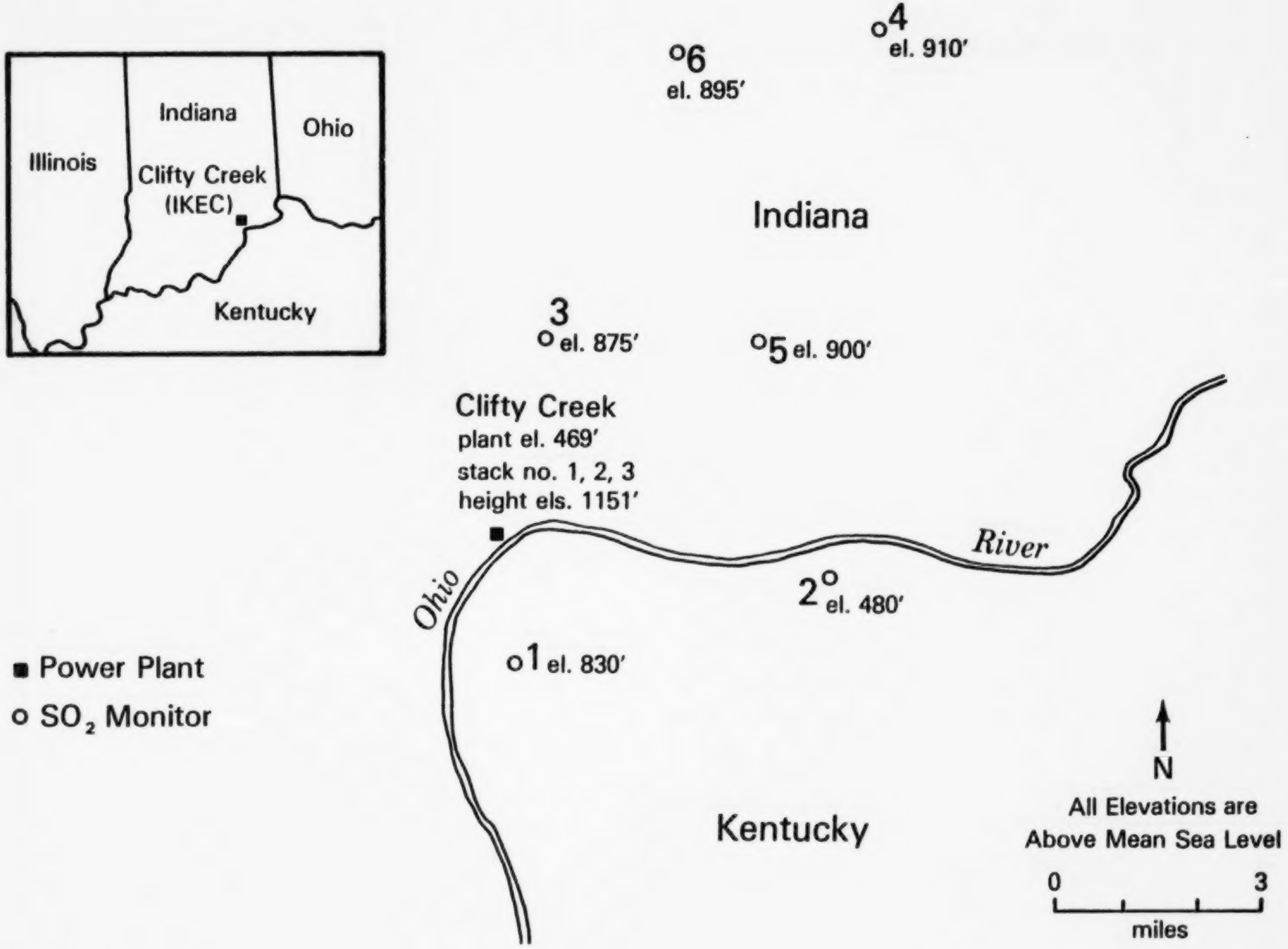
Illinois
Indiana
Ohio
Clifty Creek (IKEC)
Kentucky
4 el. 910′
6 el. 895′
Indiana
3 el. 875′
5 el. 900′
Clifty Creek
plant el. 469′
stack no. 1, 2, 3
height els. 1151′
Ohio
River
2 el. 480′
1 el. 830′
■ Power Plant
○ SO₂ Monitor
Kentucky
N
All Elevations are Above Mean Sea Level
0
3
miles

Figure VI-2. AEP monitoring network around the Clifty Creek power plant.

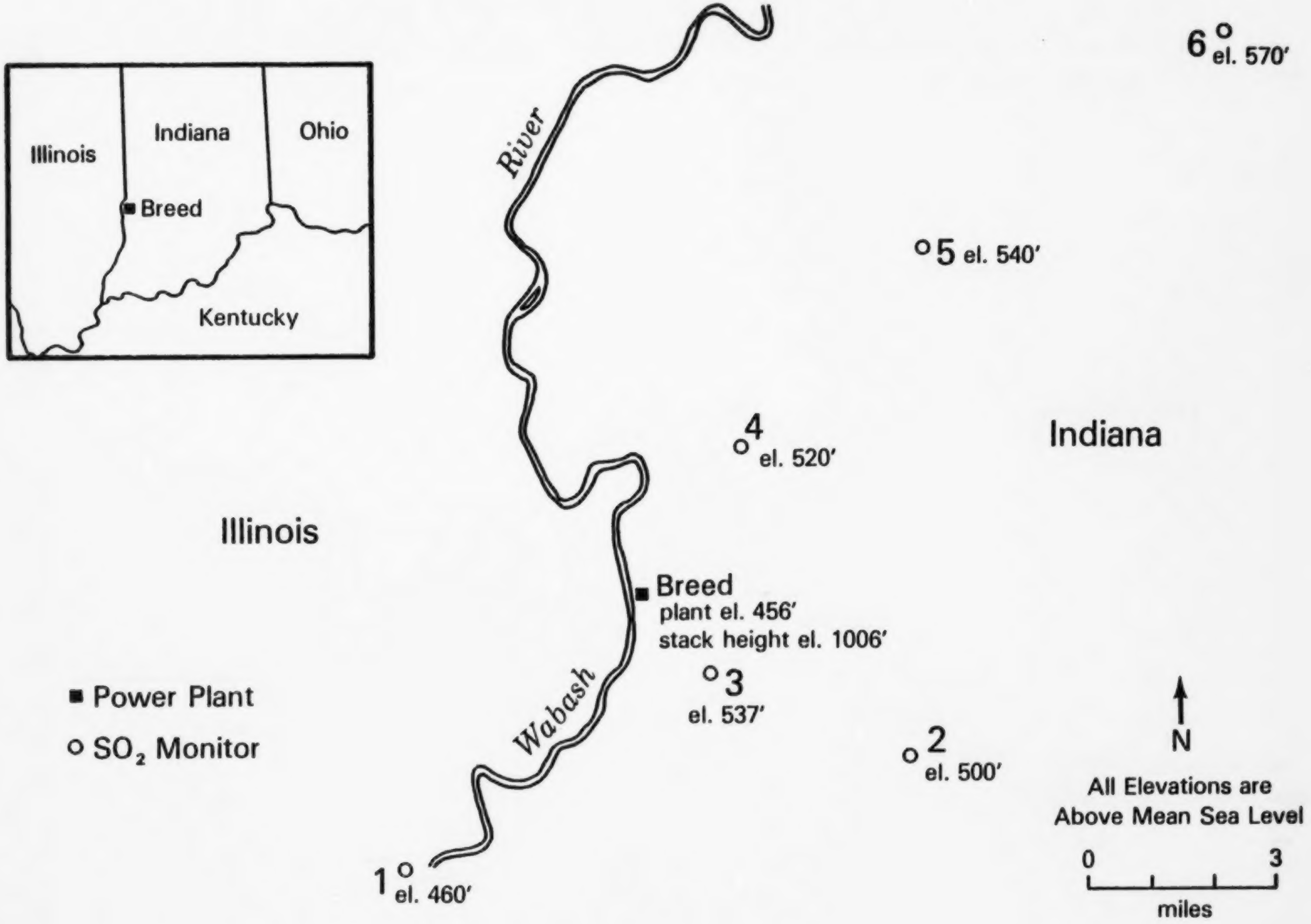
Illinois
Indiana
Ohio
Breed
Kentucky
River
Wabash
Illinois
Indiana
■ Power Plant
○ SO_2 Monitor
1 el. 460′
2 el. 500′
3 el. 537′
4 el. 520′
5 el. 540′
6 el. 570′
Breed
plant el. 456′
stack height el. 1006′
N
All Elevations are
Above Mean Sea Level
0
3
miles

Figure VI-3. AEP monitoring network around the Breed power plant.

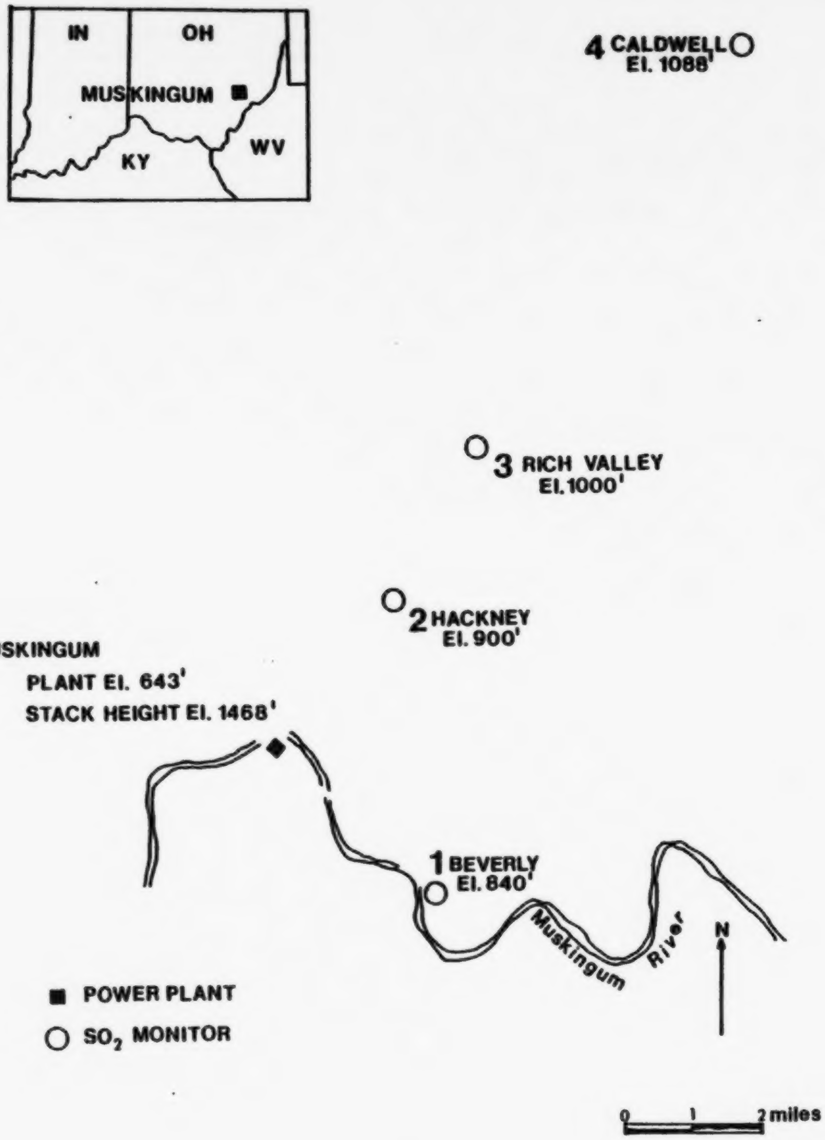
IN
OH
MUSKINGUM
KY
WV
4 CALDWELL
El. 1088'
3 RICH VALLEY
El. 1000'
2 HACKNEY
El. 900'
MUSKINGUM
PLANT El. 643'
STACK HEIGHT El. 1468'
1 BEVERLY
El. 840'
Muskingum River
N
■ POWER PLANT
○ SO_2 MONITOR
0
1
2 miles

Figure VI-4. AEP monitoring network around the Muskingum River power plant.

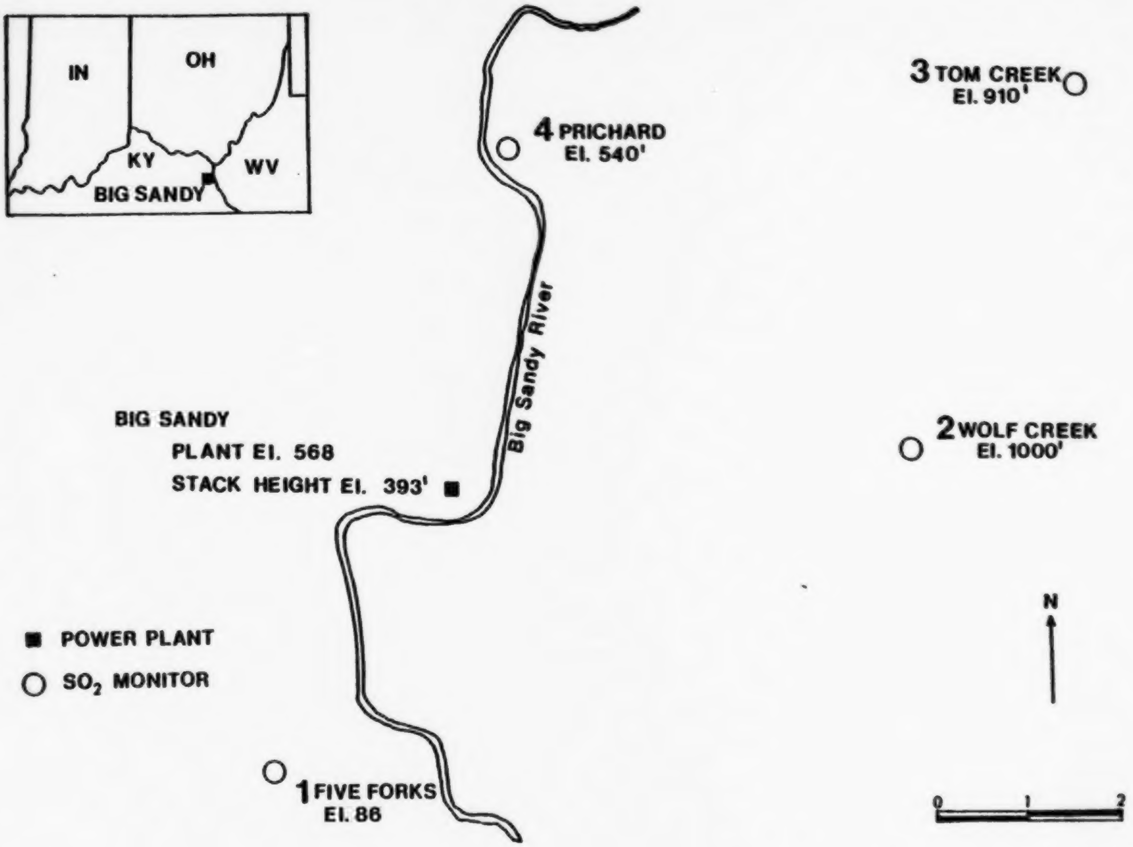
IN
OH
KY
WV
BIG SANDY
3 TOM CREEK
El. 910'
4 PRICHARD
El. 540'
Big Sandy River
BIG SANDY
PLANT El. 568
STACK HEIGHT El. 393'
2 WOLF CREEK
El. 1000'
N
■ POWER PLANT
○ SO₂ MONITOR
1 FIVE FORKS
El. 86
0
1
2

Figure VI-5. AEP monitoring network around the Big Sandy power plant.

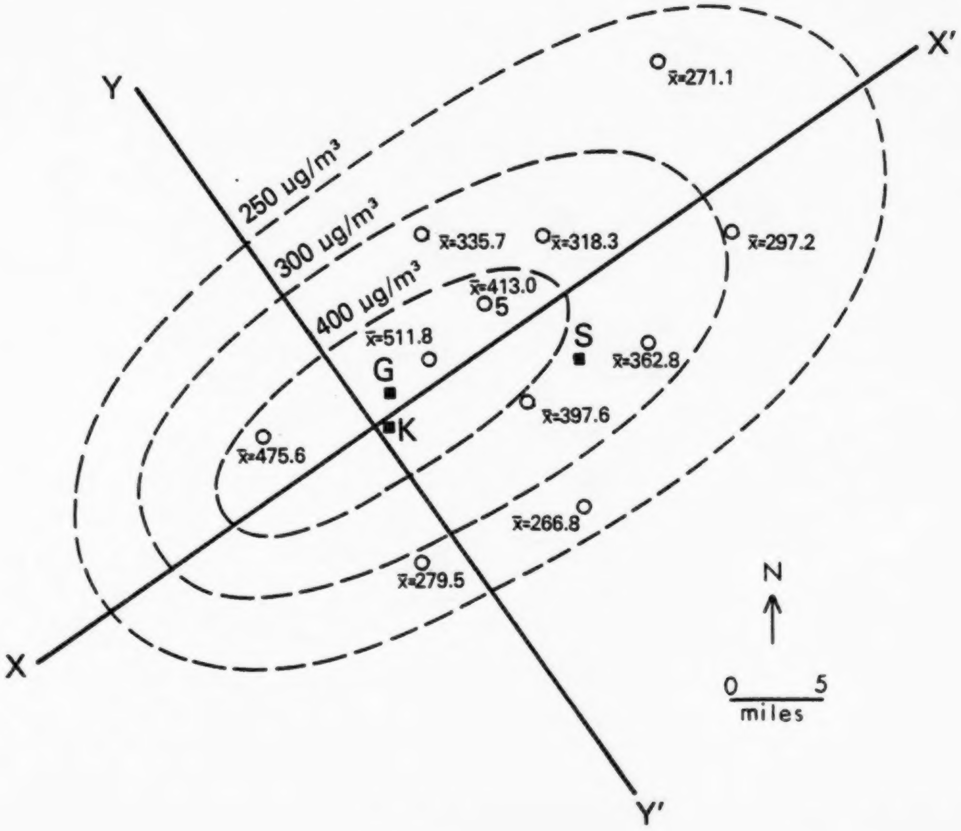
X'
Y
250 µg/m³
300 µg/m³
400 µg/m³
x̄=271.1
x̄=335.7
x̄=318.3
x̄=297.2
x̄=413.0
5
x̄=511.8
S
G
K
x̄=362.8
x̄=397.6
x̄=475.6
x̄=266.8
x̄=279.5
N
X
0
5
miles
Y'

Figure VI-6. Means of the second-highest hourly SO_2 concentrations ($\mu g/m^3$) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Gavin-Kyger Creek-Sporn power plants. These isopleths are preliminary and drawn free-hand. X-X' is downwind direction; Y-Y' is crosswind direction.

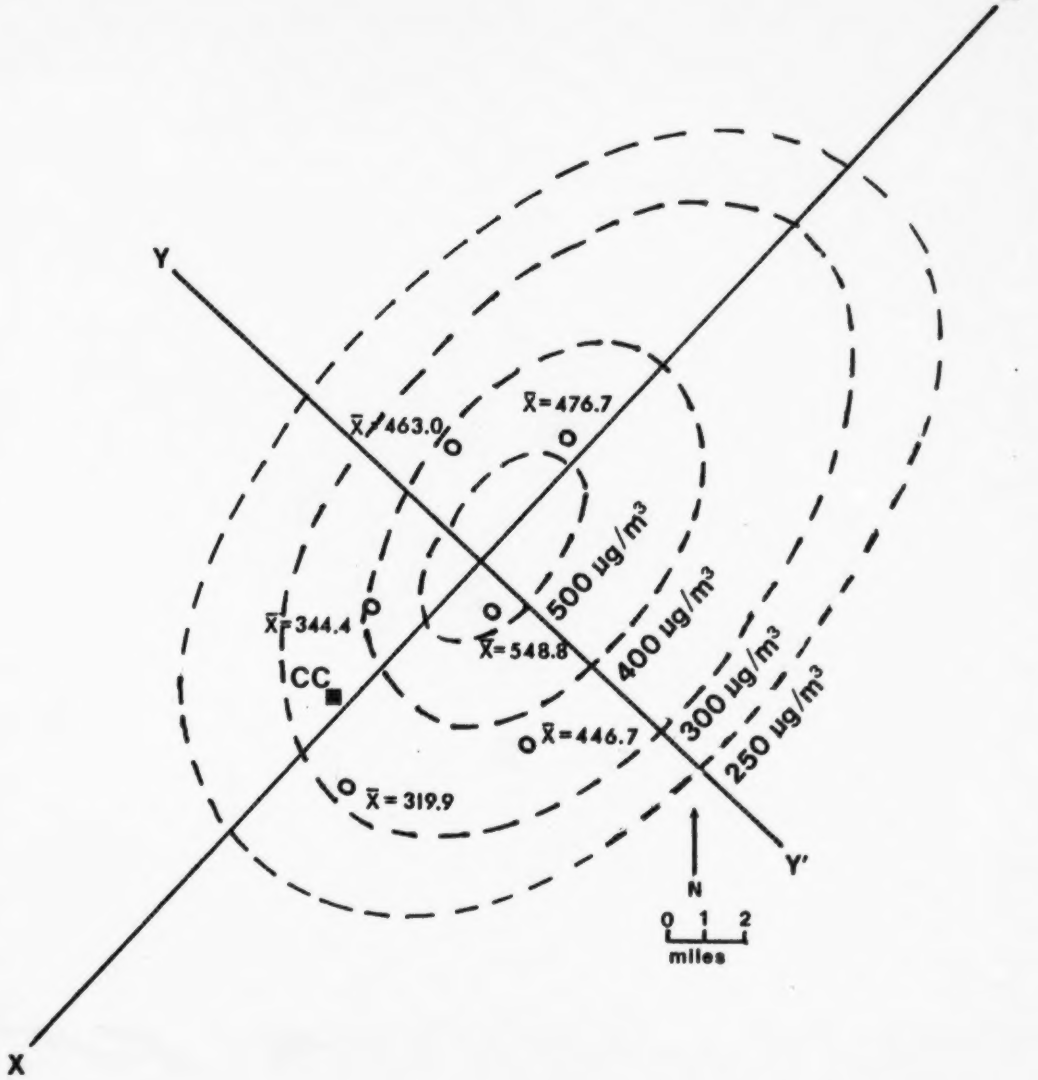
X'
Y
X̄=463.0
X̄=476.7
500 µg/m³
400 µg/m³
300 µg/m³
250 µg/m³
X̄=344.4
X̄=548.8
CC
X̄=446.7
X̄=319.9
Y'
N
0 1 2
miles
X

Figure VI-7. Means of the second-highest hourly SO_2 concentrations ($\mu g/m^3$) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Clifty Creek power plant. These isopleths are preliminary and drawn free-hand. X-X' is downwind direction; Y-Y' is crosswind direction.

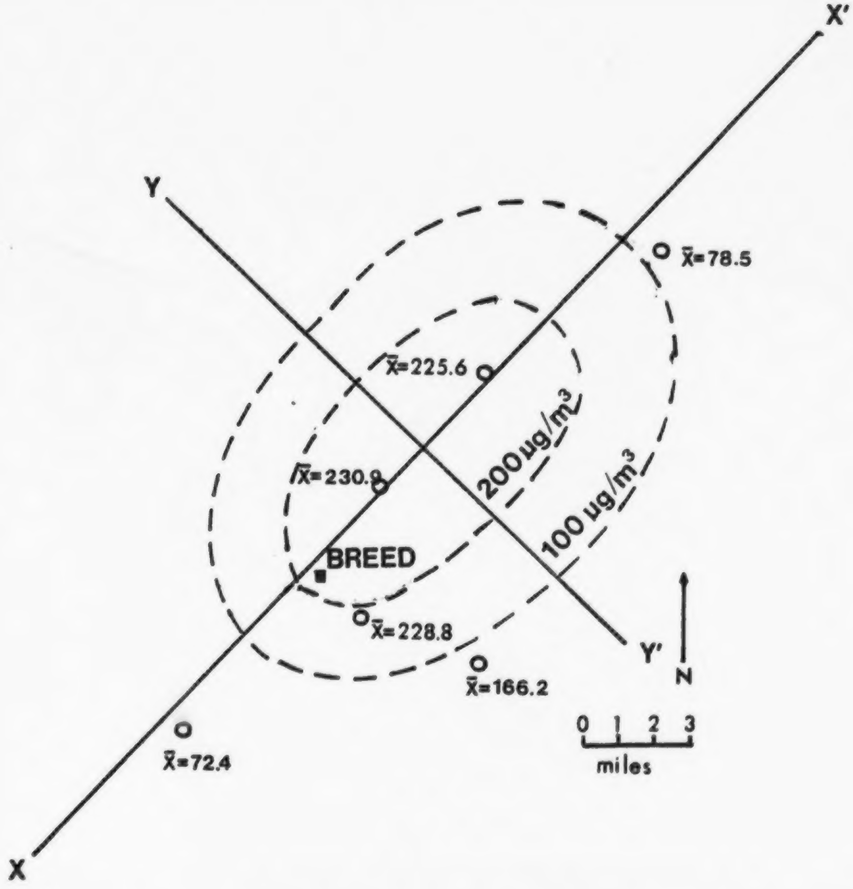
X'
Y
$\bar{X}$=78.5
$\bar{X}$=225.6
200 µg/m³
$\bar{X}$=230.8
100 µg/m³
BREED
$\bar{X}$=228.8
Y'
N
$\bar{X}$=166.2
0 1 2 3
miles
$\bar{X}$=72.4
X

Figure VI-8. Means of the second-highest hourly SO_2 concentrations (µg/m³) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Breed power plant. These isopleths are preliminary and drawn freehand. X-X' is downwind direction; Y-Y' is crosswind direction.

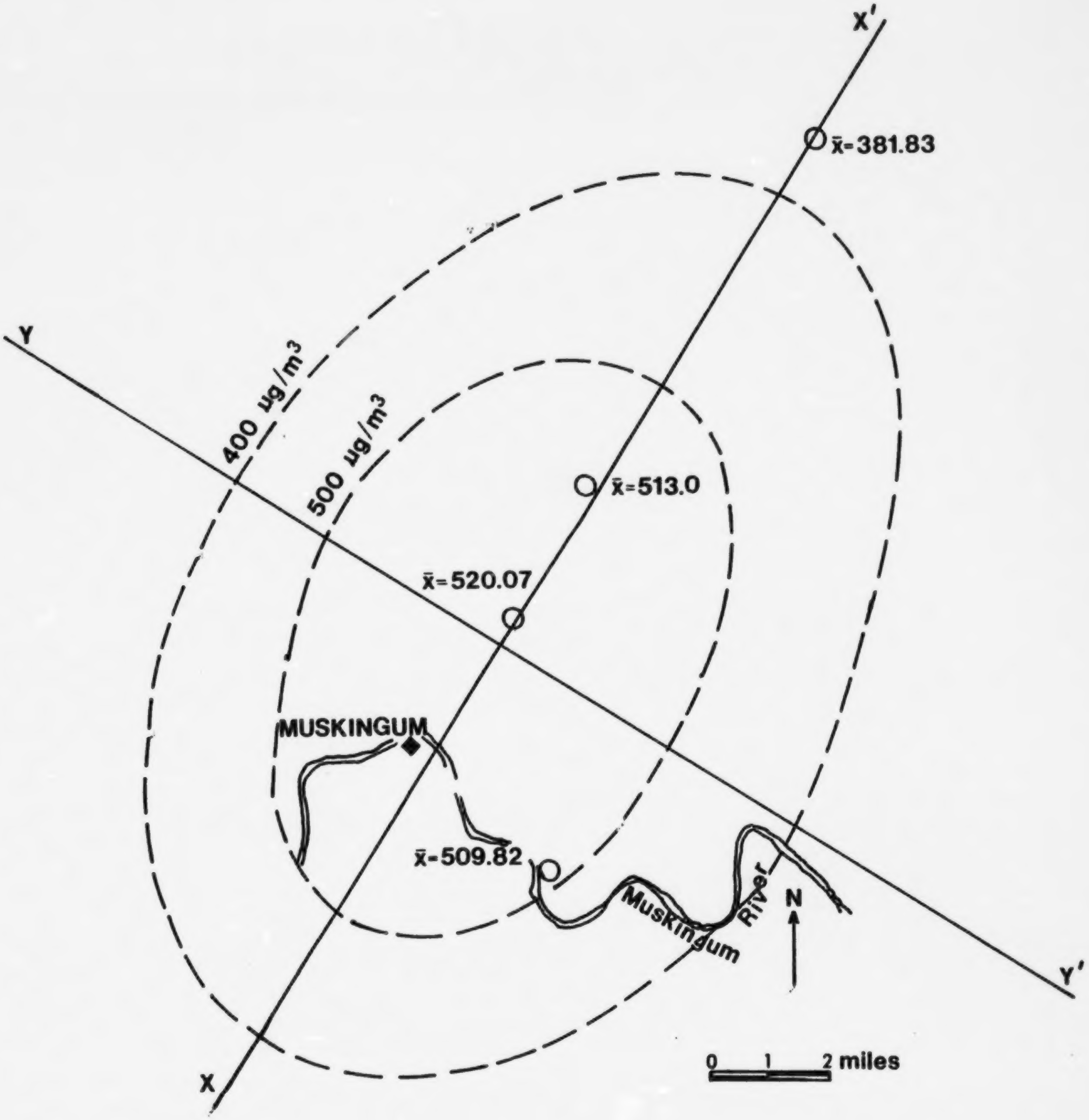
X'
x̄=381.83
Y
400 µg/m³
500 µg/m³
x̄=513.0
x̄=520.07
MUSKINGUM
x̄=509.82
Muskingum River
N
Y'
X
0 1 2 miles

Figure VI-9. Means of the second-highest hourly SO_2 concentrations ($\mu g/m^3$) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Muskingum River power plant. These isopleths are preliminary and drawn free-hand. X-X' is downwind direction; Y-Y' is crosswind direction.

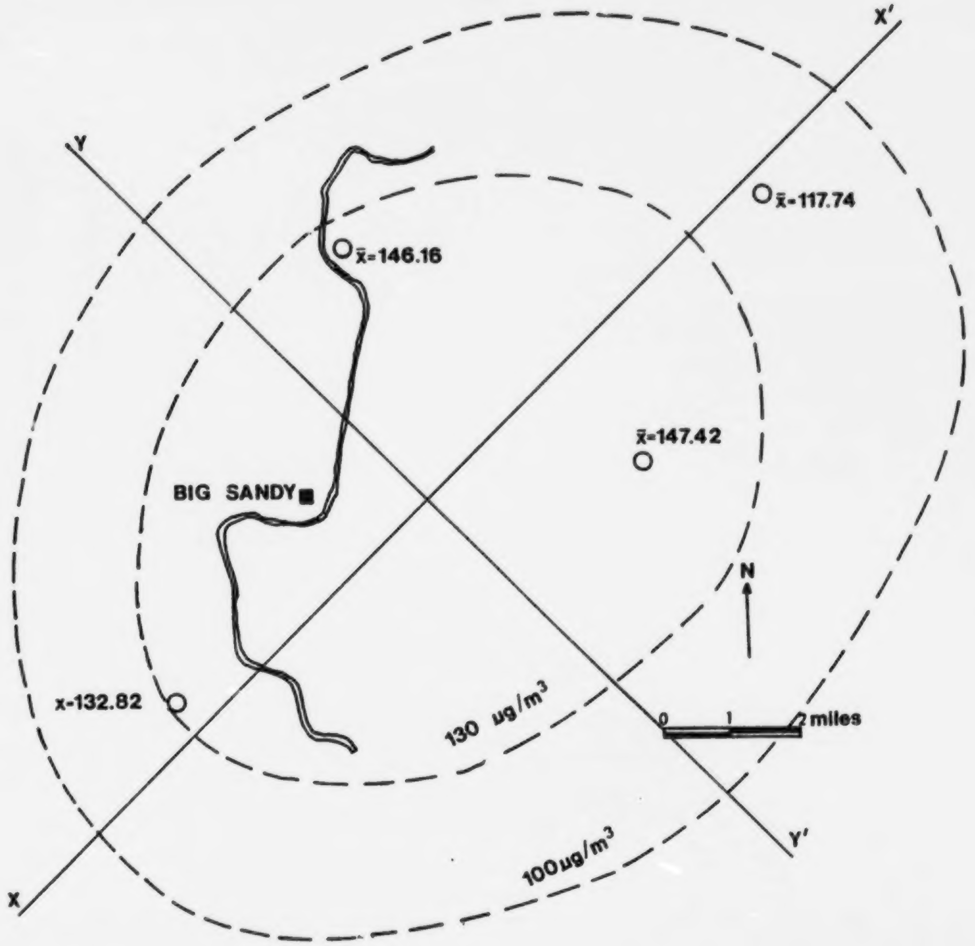
X'
Y
x̄-117.74
x̄-146.16
x̄-147.42
BIG SANDY
N
x-132.82
130 μg/m³
0
1
2 miles
Y'
100μg/m³
X

Figure VI-10. Means of the second-highest hourly SO_2 concentrations (μg/m³) during the months of June, July, August (1975-1977) for monitoring stations (o) around the Big Sandy power plant. These isopleths are preliminary and drawn free-hand. X-X' is downwind direction; Y-Y' is crosswind direction.

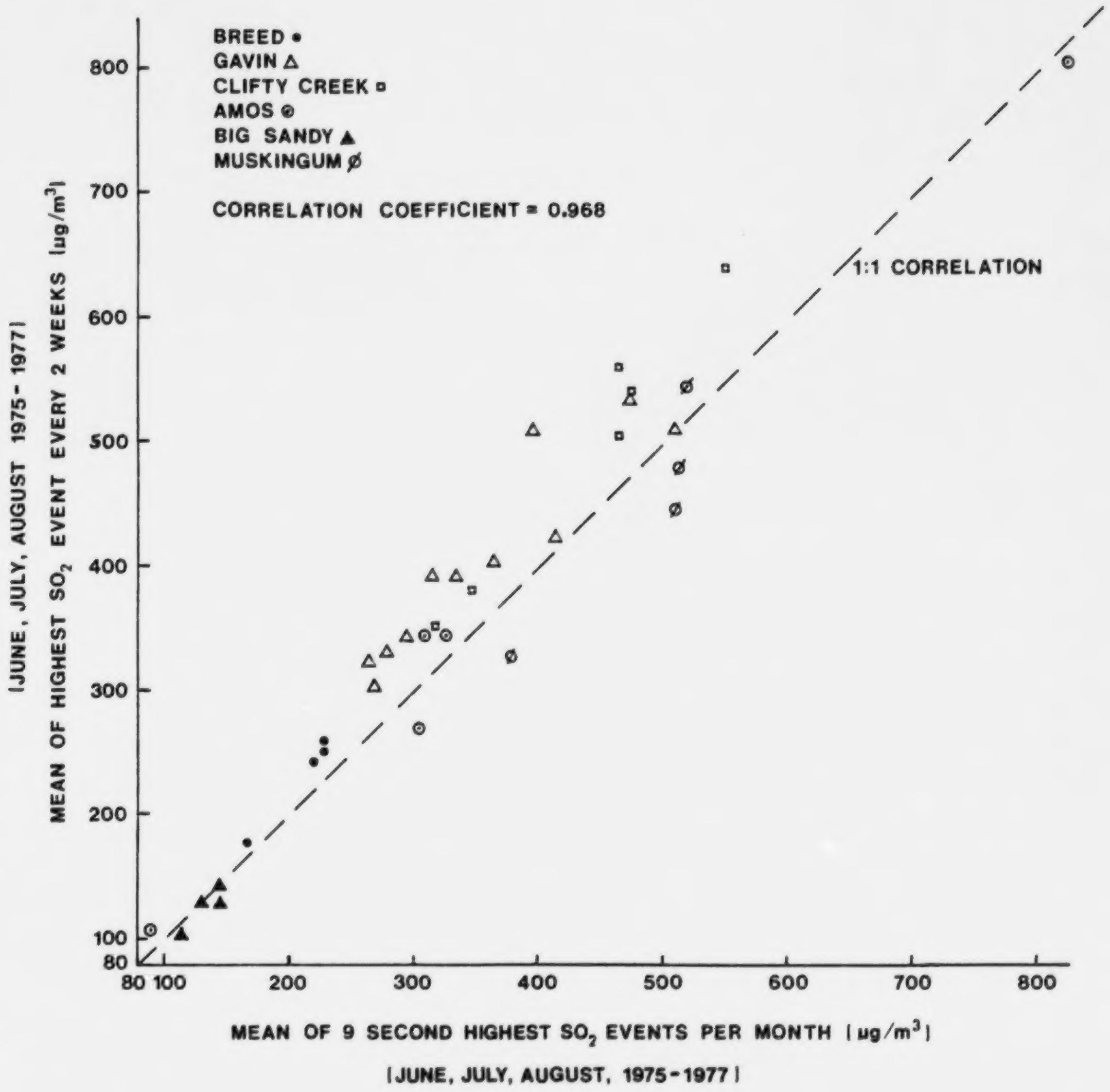

Figure VI-11. Comparison of two methods of summarizing monitored SO_2 concentrations around power plants: second-highest monthly SO_2 concentration vs. highest biweekly concentration.

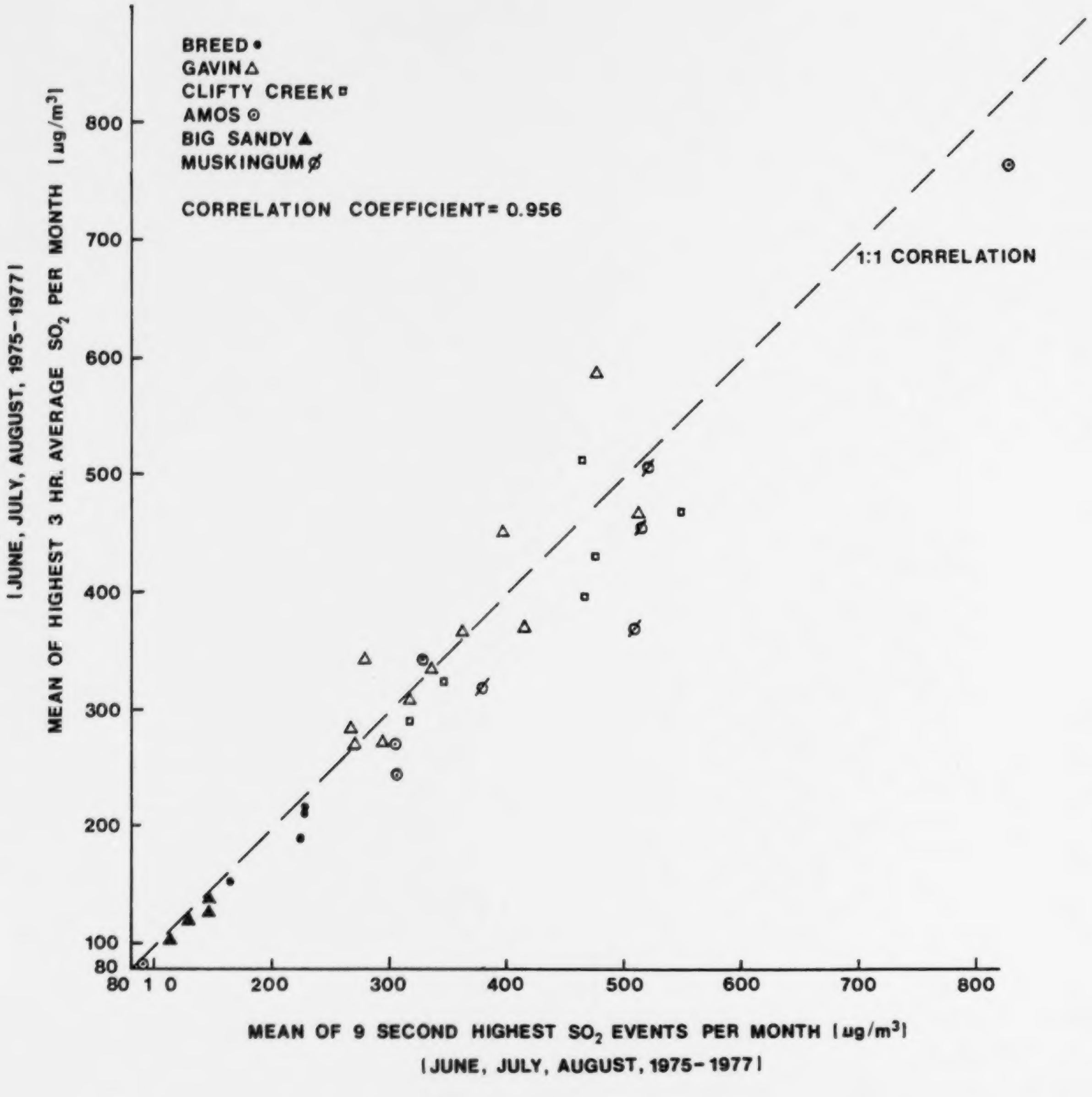

Figure VI-12. Comparison of two methods of summarizing monitored SO_2 concentrations around power plants: second-highest monthly SO_2 concentration *vs*. highest monthly three-hour average.

the network in order to extrapolate. For this purpose, the plume spread in the crosswind direction was assumed to have a Gaussian distribution with the following equation:

$$D = \alpha e^{-\beta Y^2} \qquad (1)$$

where

D = SO_2 concentration
Y = distance from origin (point of maximum concentration)
α, β = constants.

The skewed shape in the downwind direction suggests a Poisson distribution with the following equation:

$$D = CX^{\alpha} e^{-\beta x}, \qquad (2)$$

where

D = SO_2 concentration
X = distance from origin (upwind of power plant, where concentration is approximately zero)
α,β,C = constants.

To solve for the constants in Equations 1 and 2, values for D, X, and Y can be obtained from Figures VI-6 to 10 where the concentration isopleths cross the X or Y axis. For each power plant, the constants for a Gaussian and a Poisson distribution were calculated by a least-squares technique (see Appendix A). The values for D, X, and Y and the resulting constants are given in Table VI-2. Using the constants, Gaussian and Poisson curves can be drawn describing the distribution of SO_2 at the five power plants (Figures VI-13 and 14).

The curves in Figures VI-13 and 14 can be used to find the dimensions (major and minor radii) of any elliptical concentration isopleth, even those beyond the range of the monitored data. Ellipses representing the distribution of SO_2 during an average trapping event can be plotted using the following equation:

$$Y = \frac{b^2 - (X - Xm)^2 b^2}{a^2}, \qquad (3)$$

where

a = major radius (from Poisson distribution)
b = minor radius (from Gaussian distribution)
X, Y = coordinates of the ellipse
Xm = distance of center of ellipse from point of peak concentration.

TABLE VI-2. EMISSIONS, GAUSSIAN AND POISSON DISTRIBUTION CONSTANTS, AND MAJOR AND MINOR RADII FOR FIVE STUDIED OHIO RIVER BASIN POWER PLANTS

Power Plant (Peak SO_2 Emissions g/s)	Gaussian			Poisson			Major Radius(mi)		Minor Radius(mi)	
	D(μg/m³)	Y(mi)	Constants	D(μg/m³)	X(mi)	Constants	0.1 ppm	0.05 ppm	0.1 ppm	0.05 ppm
Breed	240	0	α=247.037 β=0.037195	100	15					
(3,000)	200	2.6		200	17	α=15.9278				
	100	4.9		240	22	β=0.7368	--	6.5	--	4.1
				200	27	C=1.208×10^{-12}				
				100	30					
Gavin-Kyger-Sporn	520	0		250	8.6					
	400	4	α=461.262	300	12.6					
(26,540)	300	7.7	β=5.173×10^{-3}	400	17.1	α=1.48696				
	250	11.4		520	28.3	β=0.055928	23.65	36.4	10.3	15.4
				400	40.0	C=15.5728				
				300	48.6					
				250	60.0					
Big Sandy	160	0	α=160.073	100	6.8					
(2,460)	130	4	β=1.3058×10^{-2}	130	8.8	α=2.79817				
	100	6		160	13.6	β=0.2142	--	4.7	--	3.8
				130	19.2	C=1.98376				
				100	22.0					
Muskingum River	525	0	α=532.932	400	5.0					
	500	3.2	β=8.334×10^{-3}	500	7.7	α=1.13618				
(16,130)	400	5.8		525	12.8	β=0.0984	13.5	19.75	9.1	13.0
				500	17.8	C=104.932				
				400	21.7					
Clifty Creek	510	0		300	18.0					
(9,500)	500	1.5	α=500.952	400	21.0					
	400	3.9	β=1.188×10^{-2}	500	23.5	α=8.588				
	300	6.4		545	25.5	β=0.3267	10.55	15.35	7.3	10.6
	250	7.8		500	29.0	C=1.80×10^{-6}				
				400	32.0					
				300	37.0					

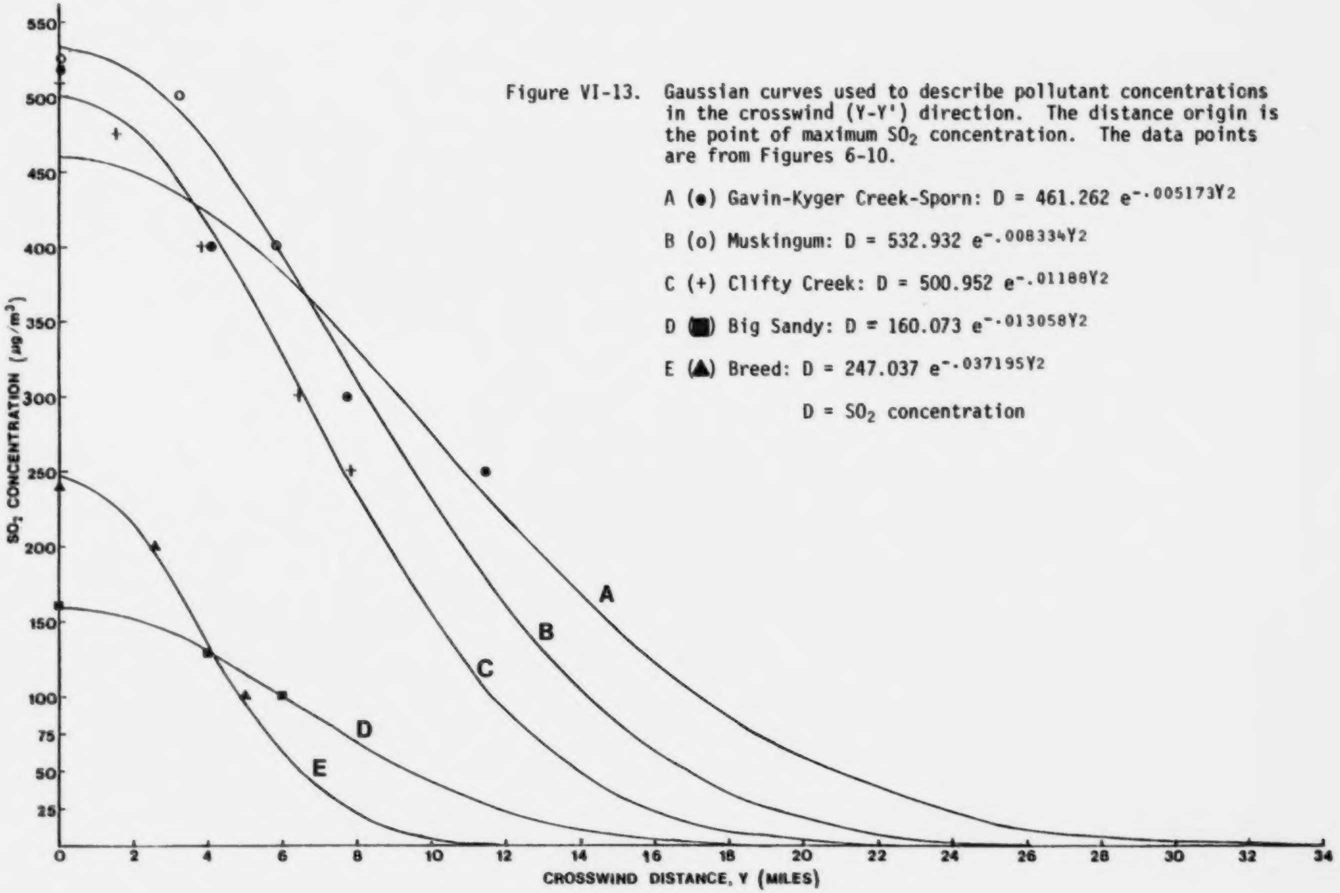

Figure VI-13. Gaussian curves used to describe pollutant concentrations in the crosswind (Y-Y') direction. The distance origin is the point of maximum SO_2 concentration. The data points are from Figures 6-10.

A (●) Gavin-Kyger Creek-Sporn: $D = 461.262\ e^{-.005173Y^2}$

B (o) Muskingum: $D = 532.932\ e^{-.008334Y^2}$

C (+) Clifty Creek: $D = 500.952\ e^{-.01188Y^2}$

D (■) Big Sandy: $D = 160.073\ e^{-.013058Y^2}$

E (▲) Breed: $D = 247.037\ e^{-.037195Y^2}$

$D = SO_2$ concentration

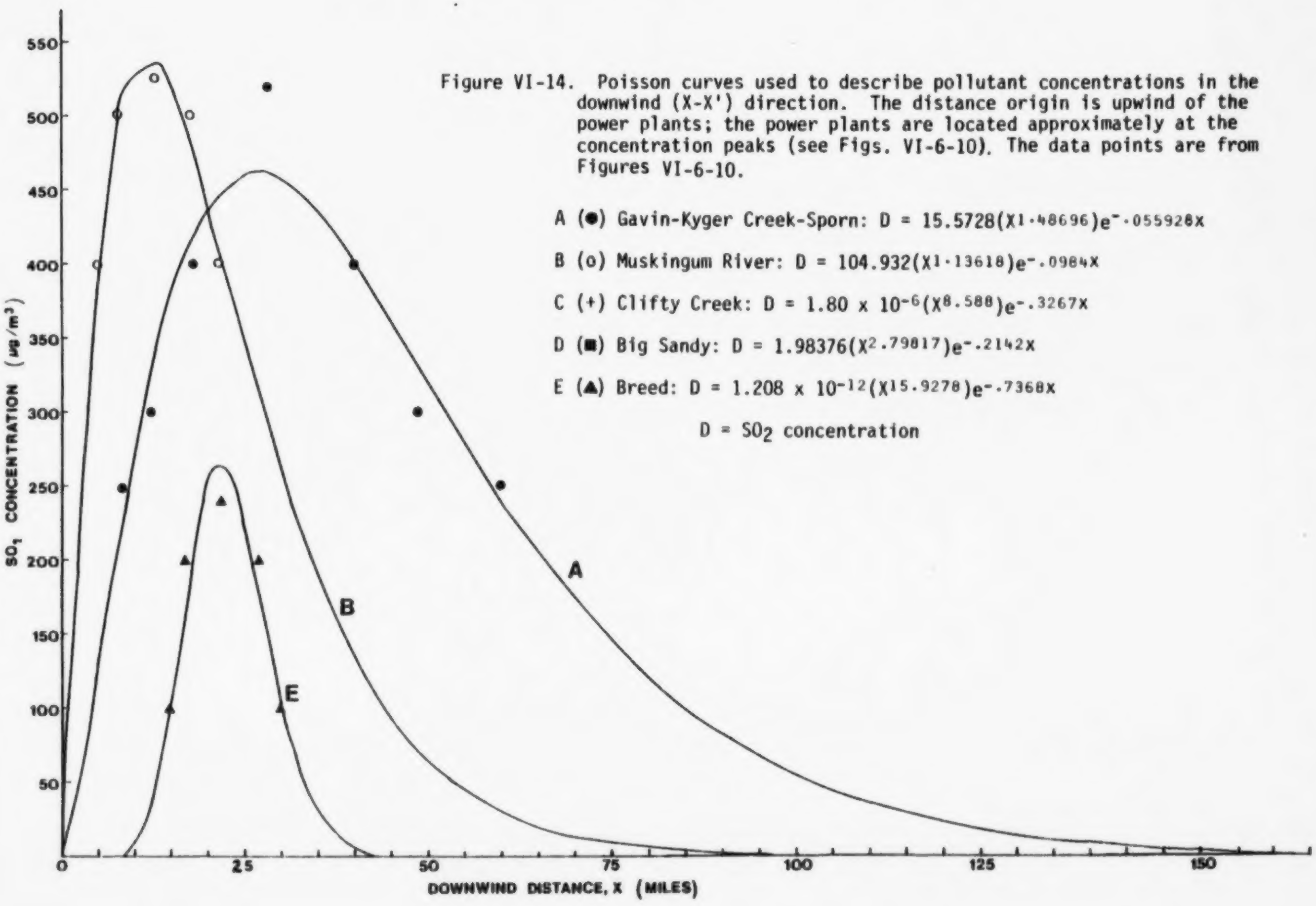

Figure VI-14. Poisson curves used to describe pollutant concentrations in the downwind (X-X') direction. The distance origin is upwind of the power plants; the power plants are located approximately at the concentration peaks (see Figs. VI-6-10). The data points are from Figures VI-6-10.

A (●) Gavin-Kyger Creek-Sporn: $D = 15.5728(X^{1.48696})e^{-.055928x}$

B (o) Muskingum River: $D = 104.932(X^{1.13618})e^{-.0984x}$

C (+) Clifty Creek: $D = 1.80 \times 10^{-6}(X^{8.588})e^{-.3267x}$

D (■) Big Sandy: $D = 1.98376(X^{2.79817})e^{-.2142x}$

E (▲) Breed: $D = 1.208 \times 10^{-12}(X^{15.9278})e^{-.7368x}$

$D = SO_2$ concentration

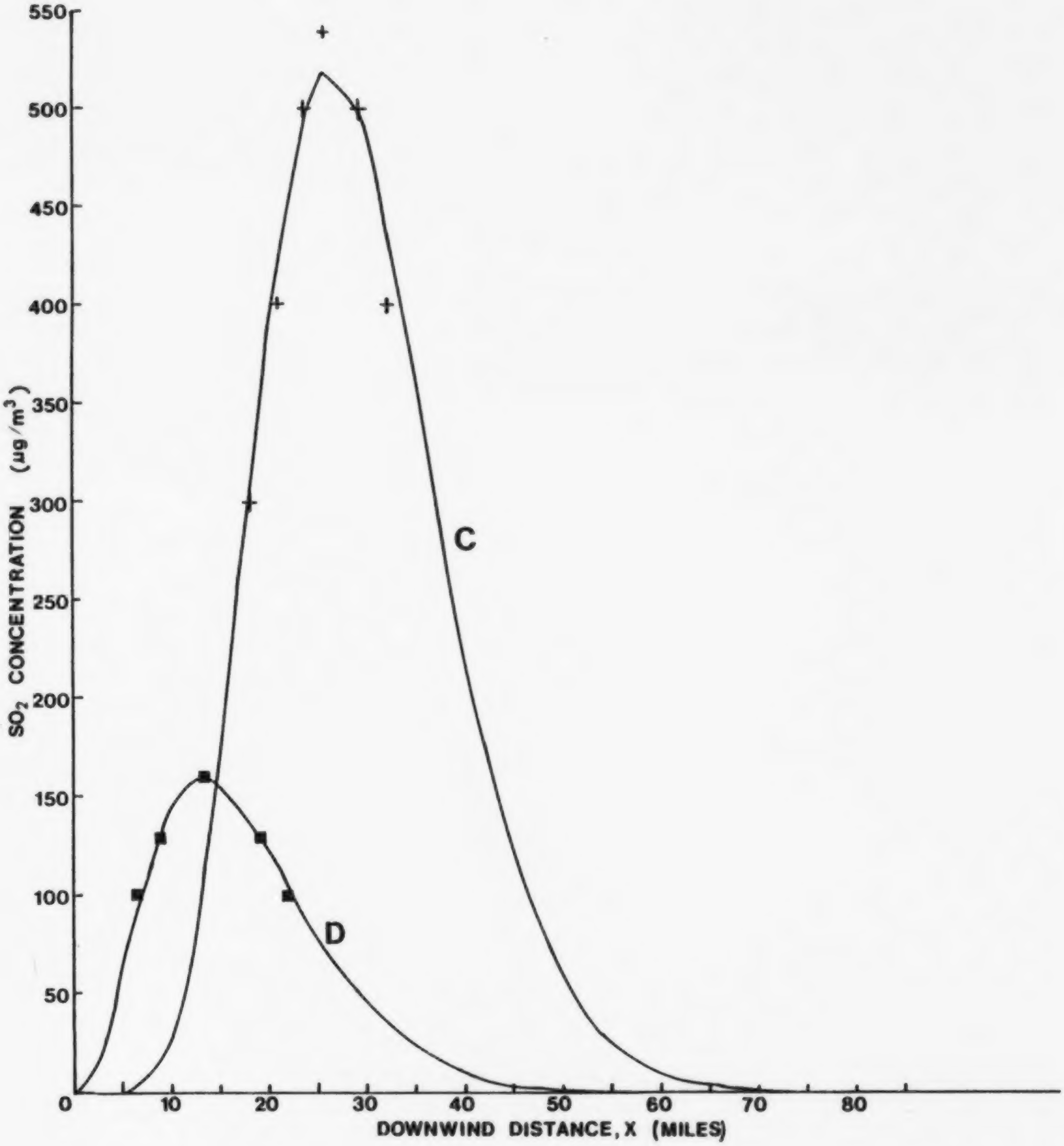

Figure VI-14. Continued

The equation for an ellipse with an arbitrary center is used because the center of each ellipse does not lie at the power plant or at the point of maximum concentration but shifts progressively downwind because of the Poisson distribution. The elliptical isopleths calculated in this way for the Gavin-Kyger Creek-Sporn complex are illustrated in Figure VI-15. The area within any elliptical isopleth is given by the equation:

$$\text{Area} = \pi ab, \tag{4}$$

where a and b are the major and minor radii.

Another method of determining concentration isopleths (impact areas) would be to let the observed concentration pattern itself suggest the distribution. Consider the concentrations, C_i, observed at locations $\vec{r}_i$, where i = 1, N = the total number of observation sites. The concentration $C(\vec{r})$ at an arbitrary point $\vec{r}$ could then be estimated by means of the following interpolation scheme:

$$C(\vec{r}) = \frac{\displaystyle\sum_{i=1}^{N} \frac{C_i}{|\vec{r} - r_i|^\alpha}}{\displaystyle\sum_{i=1}^{N} \frac{1}{|\vec{r} - \vec{r}_i|^\alpha}} \tag{5}$$

where the weighting factor $\frac{1}{|\vec{r} - \vec{r}_i|^\alpha}$ reflects the fact that the correlation between two concentrations will be generally reduced as the separation between the monitoring sites is increased. A weighting factor of the form

$$\frac{e^{-\beta|\vec{r} - \vec{r}_i|}}{|\vec{r} - \vec{r}_i|^\alpha}$$

could also have been used. The best value for α in equation 5 could be obtained through an optimization procedure using the actual observed concentrations in the left hand side of equation 5:

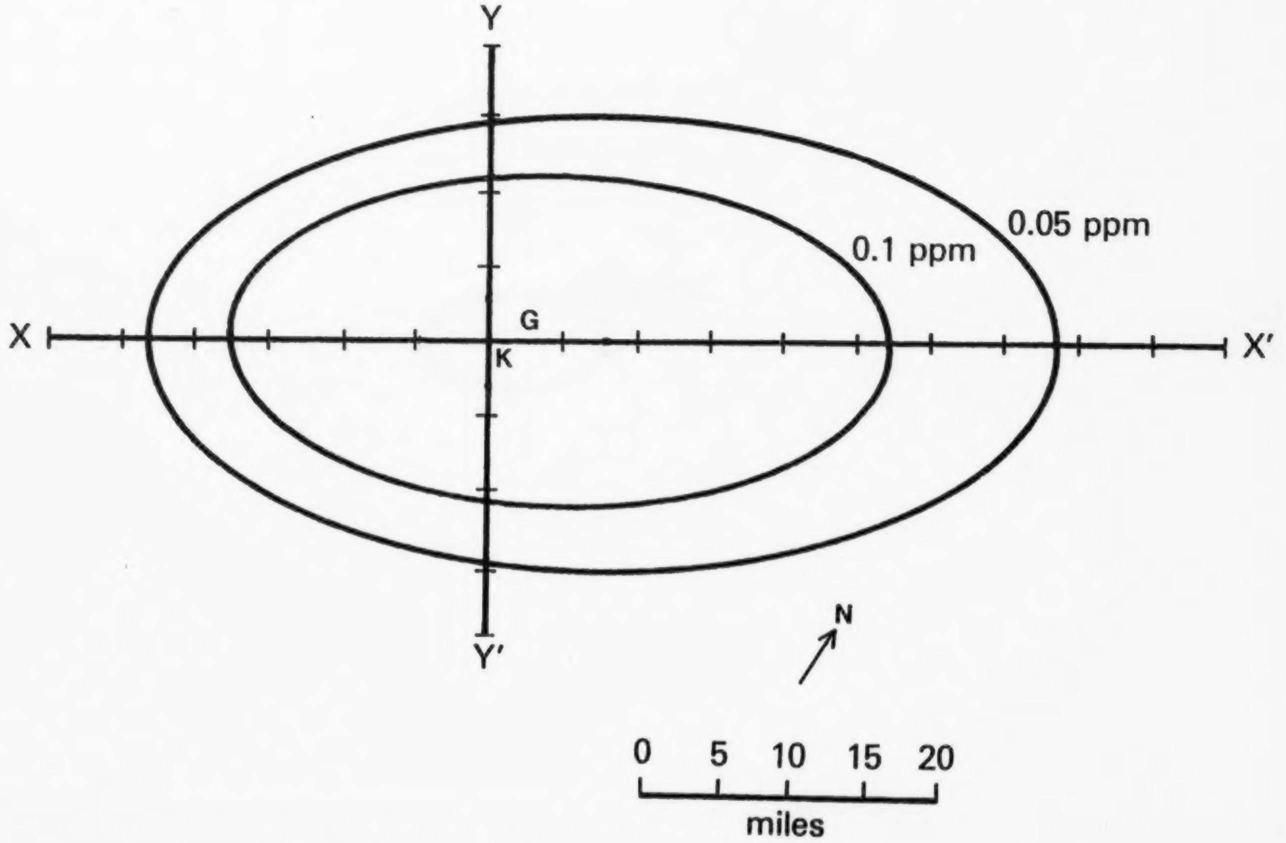
Y
Y'
X
X'
G
K
0.1 ppm
0.05 ppm
N
0
5
10
15
20
miles

Figure VI-15. Calculated SO_2 concentration isopleths for the Gavin (G)-Kyger Creek (K)-Sporn power plants (see text). Compare to Figure VI-6.

$$C_j(\vec{r}_j) = \frac{\sum\limits_{\substack{i=1\\ i\neq j}}^{N} \frac{C_i}{|\vec{r} - \vec{r}_i|^{\alpha}}}{\sum\limits_{\substack{i=1\\ i\neq j}}^{N} \frac{1}{|\vec{r} - \vec{r}_i|^{\alpha}}}$$

Obviously the accuracy of this interpolation procedure would be greatest for those points within the area enclosed by the actual measurement locations; estimates outside this area would be considerably less accurate. The location of most of the SO_2 monitors to the northeast of each plant would lead to an underestimation of the area impacted by the plant. This problem could be alleviated somewhat by use of wind direction data to determine the frequency of similar impacts in other directions.

SO_2 CONCENTRATION ISOPLETHS AROUND POWER PLANTS WITHOUT MONITORING DATA

In order to determine the area within concentration isopleths at other power plants, the major and minor radii of the 0.1-ppm and 0.05-ppm isopleths for the five sites studied in detail (Table VI-2) are plotted against peak load SO_2 emission rates in Figures VI-16 and 17. The size of the major and minor radii are taken from Figures VI-13 and 14 at the appropriate concentrations. The peak load emission rates are from Teknekron, Inc. (1980). Using these graphs, the major and minor radii can be calculated for the 0.1-ppm and 0.05-ppm isopleths around any power plant, given only the peak emission rate. The area within each isopleth is calculated with Equation 4. Peak load emissions are used because the high summer SO_2 concentrations used to define the isopleths are most probably a product of these high emission rates.

The relationship between emission rate and the major radius (Figure VI-16) is linear over the range of emissions studied but must be nonlinear at low emission rates. For example, the Breed power plant has an emission rate of 3000 g SO_2/sec, but fails to produce a monitored concentration of 0.1 ppm (260 $\mu g/m^3$) (see Figure VI-8). For this reason, power plants with emissions less than 3200 g/sec are assumed to produce no concentrations of SO_2 above 0.1 ppm. The cutoff point for the 0.05 ppm isopleth was set at 2000 g/sec; emissions less than this appear to produce a negligibly small impact area, in terms of peak concentrations twice per month.

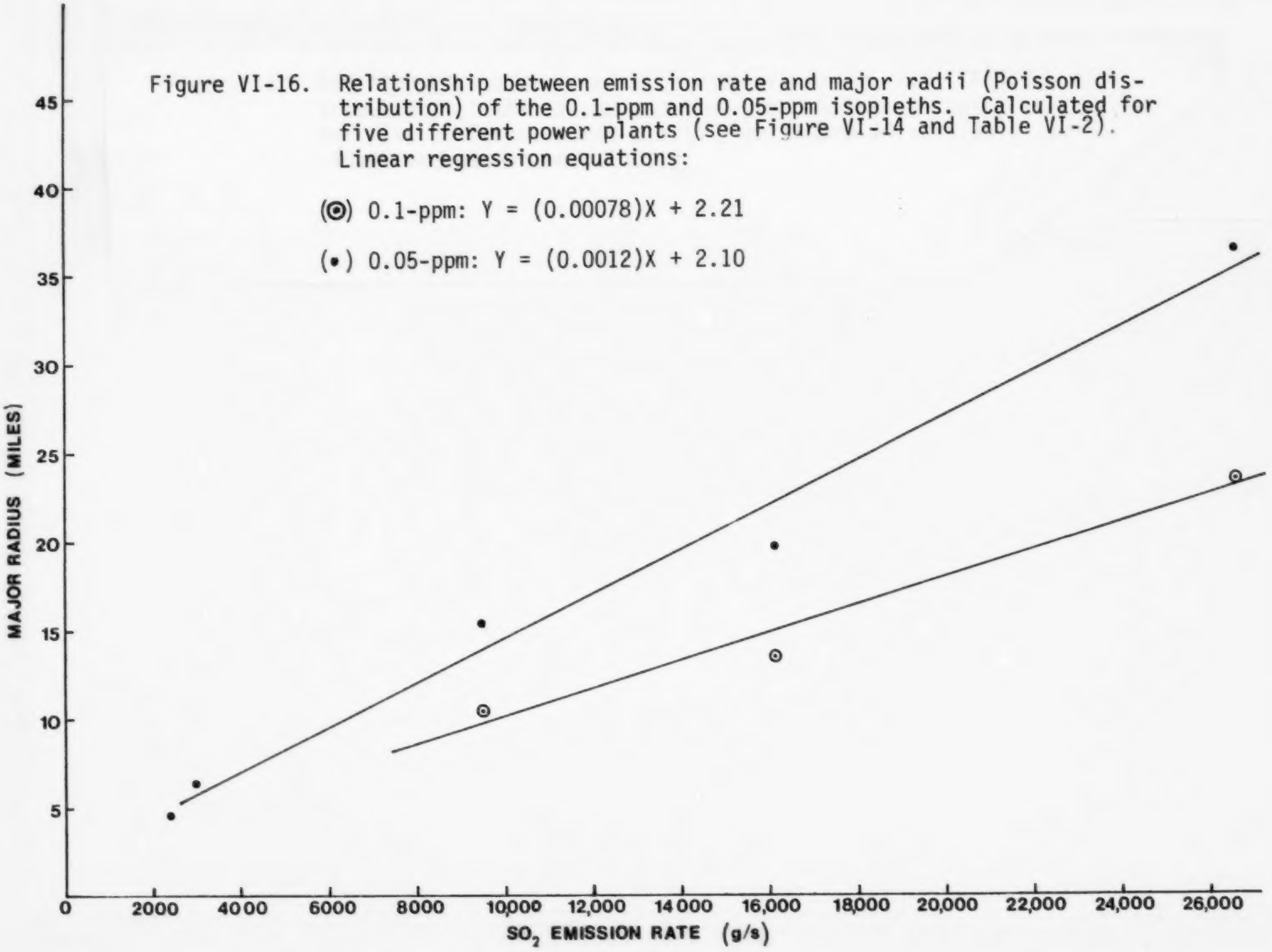

Figure VI-16. Relationship between emission rate and major radii (Poisson distribution) of the 0.1-ppm and 0.05-ppm isopleths. Calculated for five different power plants (see Figure VI-14 and Table VI-2). Linear regression equations:

(◎) 0.1-ppm: $Y = (0.00078)X + 2.21$

(•) 0.05-ppm: $Y = (0.0012)X + 2.10$

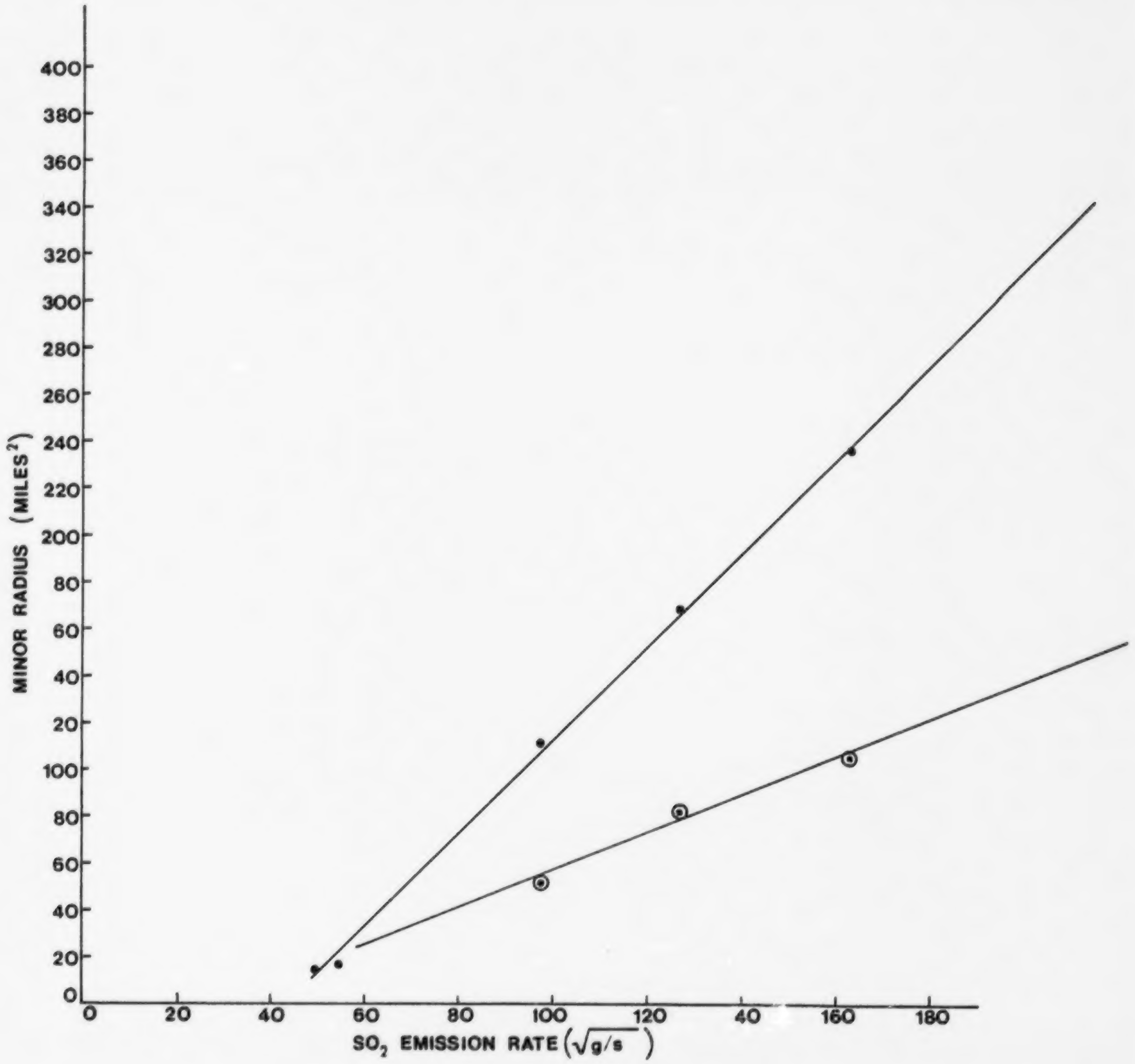

Figure VI-17. Relationship between emission rate and minor radii (Gaussian distribution) of the 0.1-ppm and 0.05-ppm isopleths. Calculated for five different power plants (see Figure VI-13 and Table VI-2). Linear regression equations:

(◉) 0.1-ppm: Y = (0.8)X - 22.6

(•) 0.05-ppm: Y = (2.0)X - 87.7

In addition to the above considerations, the relationship between emissions and the minor radius was linear only with the specified data transformation (Figure VI-17). Several transformations were attempted, and the form used here gave the best linear correlation coefficient. The reasons for this relationship are not clear, but the overall effect is that the minor radius expands more slowly than the major radius as the emission rate increases. Smaller power plants have nearly circular isopleths, while larger plants have progressively more elongated isopleths. This may be a real effect in which the larger amounts of SO_2 from larger power plants are not scavenged by vegetation or transformed and are therefore transported a greater distance downwind. The emissions from smaller power plants may not persist long enough to be transported significant distances by the wind. This phenomenon also may be partially due to the placement of monitors and the subsequent construction of isopleths. Most monitors are downwind (northeast) of the plants. Relatively fewer are placed to detect crosswind variation in concentration, possibly leading to an underestimate of the crosswind spread of pollutants.

Besides emission rate, another possible factor affecting the size, shape, and position of the isopleths is the stack height of the plant. In theory, a higher stack should minimize the concentration of pollutants on the ground, thereby decreasing the area of the isopleths. Higher stacks also should cause the point of maximum pollutant concentration to be shifted downwind of the plant. In actual fact, these relationships are not clear-cut. The point of maximum pollutant concentration at Gavin-Kyger Creek-Sporn, which has the highest stack (Table VI-3), is actually slightly upwind of the plants (Figure VI-6). At Clifty Creek, with the lowest effective stack height (Table VI-3), the maximum SO_2 concentration is about 6 miles downwind (Figure VI-7), the greatest displacement of any plant. The relationship between effective stack height and isopleth area is not straightforward either (Table VI-3). The plant with the highest stack (Gavin-Kyger Creek-Sporn) has the greatest isopleth area because of its high emission rate. Big Sandy has the highest stack relative to emissions and the smallest, most circular isopleth area, which is adequately predicted by the emission rate alone (Figures VI-16 and 17). Clifty Creek's isopleth area, relative to emissions, is somewhat larger than expected, which could be due to its low effective stack height, but, again, the area is predicted well by emission rate alone.

Two conclusions seem apparent: First, not enough data are yet available to describe completely the effect of stack height on the area, shape, or position of impact areas; and, second, the effect of stack height on isopleth area seems to be much smaller than that of emission rate. This latter conclusion is not surprising in view of the apparent meteorology of the trapping events. A high stack should have little effect when pollutants are trapped in a stagnant air mass under a stable layer of air.

Using the standard relationships between peak emissions rates and isopleth radii (from Figures VI-16 and 17), the impact areas for all power plants with peak SO_2 emission rates of 2000 grams per second (g/sec) or

TABLE VI-3. STACK HEIGHTS AND ISOPLETH AREAS OF FIVE STUDIED OHIO RIVER BASIN POWER PLANTS

Power Plant (Peak SO_2 emissions g/s)	Actual Stack Height (ft)	Effective Stack Height (ft)[a]	Isopleth Area (acres)	
			0.1 ppm	0.05 ppm
Big Sandy (2,460)	825	566	--	35,910
Breed (3,000)	550	485	--	53,583
Clifty Creek (9,500)	682	336	154,848	327,148
Muskingum (16,130)	825	523	247,005	516,227
Gavin-Kyger-Sporn (26,540)	879[b]	678[b]	489,777	1,127,073

[a] The effective stack height is the difference between the top of the stack and the average elevation of the monitors

[b] Emission-weighted average for the three plants

higher were calculated for 1976 (Table VI-4). A map of the distribution of these impact areas is shown as Figure VI-19.

The emission associated with the plants required under three development scenarios through 1985 and 2000 are shown in Figure VI-18. Scenarios 2 and 2d project a doubling of generating capacity between 1976 and 2000. Scenario 7 projects a roughly 15% higher generating capacity in 2000 than does Scenario 2, accomplished partially by extending power plant years in service from 35 to 45 years. The differences between the three scenarios produce the total emission changes shown in Figure VI-18. The development should be viewed as three options and three dates:

(1) 1976 peak emissions of power plants in the ORBES region (Table VI-4, Figure VI-19), calculated by the AIRTEST submodel of the Utility Simulation Model (USM) and based on actual coal usage.

(2) 1985 projected peak emissions of power plants in the ORBES region under three scenarios of generating capacity (projected by the USM but based on different assumptions)(Teknekron 1980):

    a) Scenario 2, assuming compliance with State Implementation Plans (SIPs) and doubling of generating capacity (see Figure VI-20 for a map of the impact areas).

    b) Scenario 2d, assuming no compliance with SIPs and doubling of generating capacity (see Figure VI-21 for the impact areas).

    c) Scenario 7, assuming compliance with SIPs and high electrical growth (see Figure VI-22 for the impact areas).

(3) 2000 projected peak emissions of power plants in the ORBES region under three scenarios of generating capacity (projected by the USM but based on different assumptions):

    a) Scenario 2, assuming compliance with SIPs and doubling of generating capacity (see Figure VI-23 for the impact areas).

    b) Scenario 2d, assuming no compliance with SIPs and doubling of generating capacity (see Figure VI-24 for the impact areas).

    c) Scenario 7, assuming compliance with SIPs and high electrical growth (see Figure VI-25 for the impact areas).

Complete listings of the emissions and projected impact areas for 1985 and the year 2000 are available on file. The emissions on which the calculations are based were supplied by Teknekron, Inc. (1980). Power plants in close proximity were treated as a single source. The error

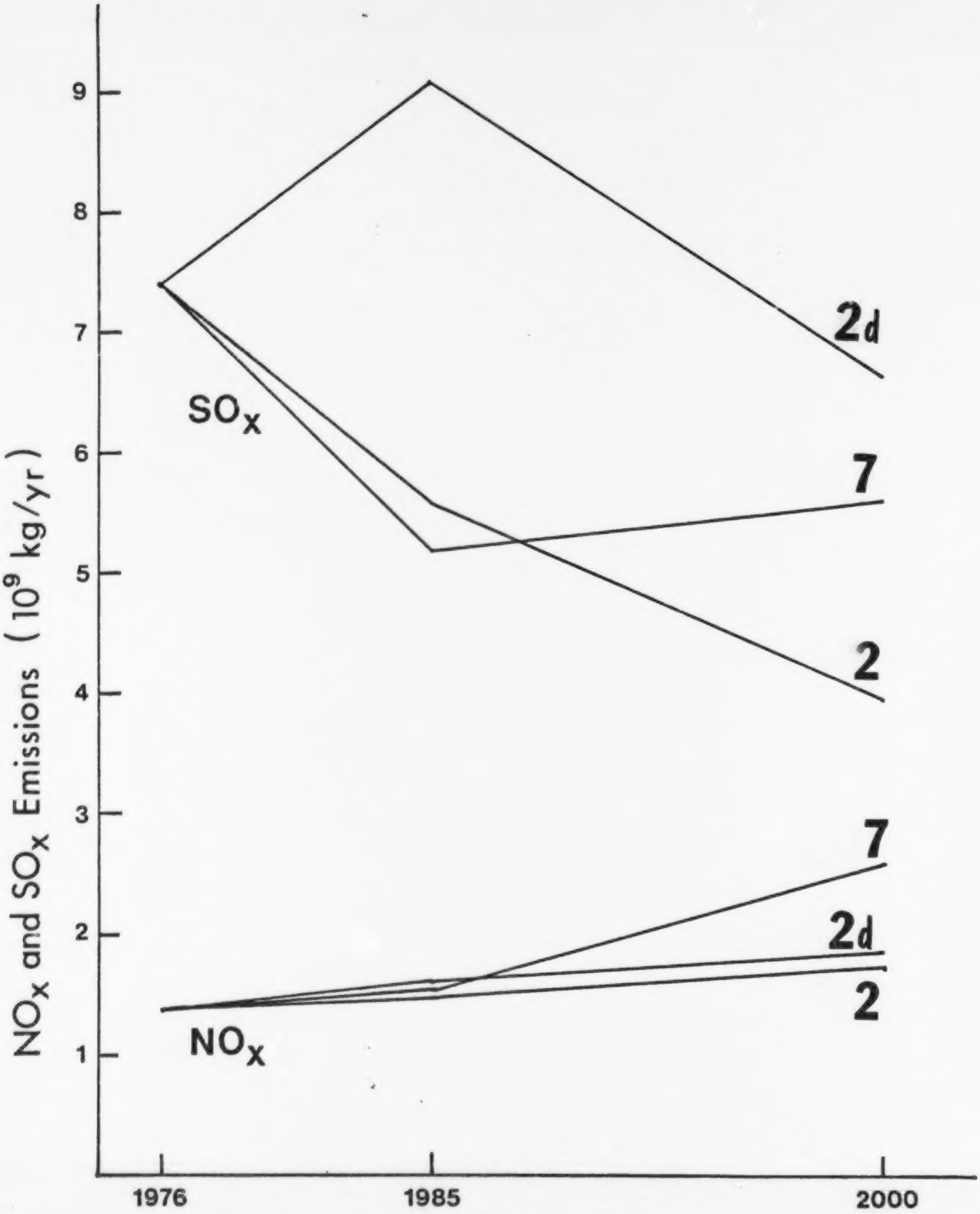

Figure VI-18. Projections of nominal NO_x and SO_x emissions (1980-2000) under three scenarios of utility development in the Ohio River Basin.

TABLE VI-4. SULFUR OXIDE EMISSIONS AND CALCULATED CONCENTRATION ISOPLETHS OF POWER PLANTS IN THE ORBES REGION FOR 1976. (Included are power plants with emissions greater than 2000 g/sec.)

			Area Inside Calculated Concentration Isopleth (Acres)	
	Power Plant	SO_2 Emissions (g/sec)	0.05 ppm	0.1 ppm
IL	Kincaid	14,760	496,384	238,298
	Joppa	5,640	140,960	81,354
	Coffeen	9,540	282,620	144,613
	Edwards	3,560	72,055	50,266
	Baldwin	12,450	398,752	195,698
	Powerton	15,240	517,222	247,359
	Meredosia	2,510	36,339	--
	Hennepin	2,460	34,441	--
	Duck Creek	2,870	49,153	--
State	Totals	69,030	2,027,926	957,588
IN	Gibson	4,580	105,427	65,294
	Clifty Creek	9,500	281,082	143,933
	Cayuga	5,590	139,257	80,585
	Culley and Warrick	9,500	281,082	143,933
	Gallagher	5,170	125,060	74,173
	Petersburg and Ratts	6,540	172,108	95,376
	Stout	3,960	85,101	56,116
	Tanner's Creek	7,760	215,833	114,948
	Wabash River	4,860	114,702	69,489
	Breed	3,000	53,564	--
	Pritchard	2,080	17,323	--
State	Totals	62,540	1,590,539	843,847
KY	Ghent	4,100	89,672	58,175
	Big Sandy	2,460	34,441	--
	Brown	2,570	38,566	--
	Cane Run	10,840	333,492	167,057
	Elmer Smith	3,500	70,095	49,392
	Henderson II	6,840	182,700	100,129
	Mill Creek	5,840	147,800	84,439
	Paddy's Run	3,620	74,014	51,140
	Paradise	25,890	1,020,375	464,049
	Shawnee	12,700	409,098	200,225
	Coleman	4,520	103,448	64,399
State	Totals	82,880	2,503,701	1,239,005

OH	Stuart	9,090	265,413	136,994
	Burger	4,480	102,131	63,803
	Beckjord	5,920	150,550	85,678
	Conesville	17,220	605,043	285,445
	Miami Fort	4,870	115,035	69,640
	Cardinal and Tidd	8,120	229,061	120,844
	Sammis	16,510	573,215	271,659
	Muskingum R.	16,130	556,332	264,339
	Poston	2,410	32,498	--
	Philo	3,560	72,055	50,266
	Gavin, Kyger and Sporn	26,540	1,053,285	478,127
State Totals		114,850	3,754,618	1,826,795
PA	Conemaugh	8,000	224,635	118,872
	Elrama	2,020	13,457	--
	Hatfield's Ferry	6,920	185,542	101,403
	Homer City	6,680	177,038	97,589
	Keystone	7,100	191,965	104,280
	Mansfield	4,920	116,700	70,393
	Mitchell	2,390	31,706	--
	New Castle	2,670	42,180	--
	Phillips	2,230	24,941	--
	Shawville	2,680	42,536	--
	Cheswick	2,550	37,829	--
State Totals		48,160	1,093,529	492,537
WV	Amos	5,440	134,164	78,286
	Fort Martin	4,490	102,460	63,952
	Harrison	9,810	293,046	149,222
	Kammer and Mitchell	14,440	482,594	232,296
	Mount Storm	6,460	169,301	94,115
State Totals		40,640	1,181,565	617,871
ORBES TOTALS		418,100	12,151,878	5,977,643

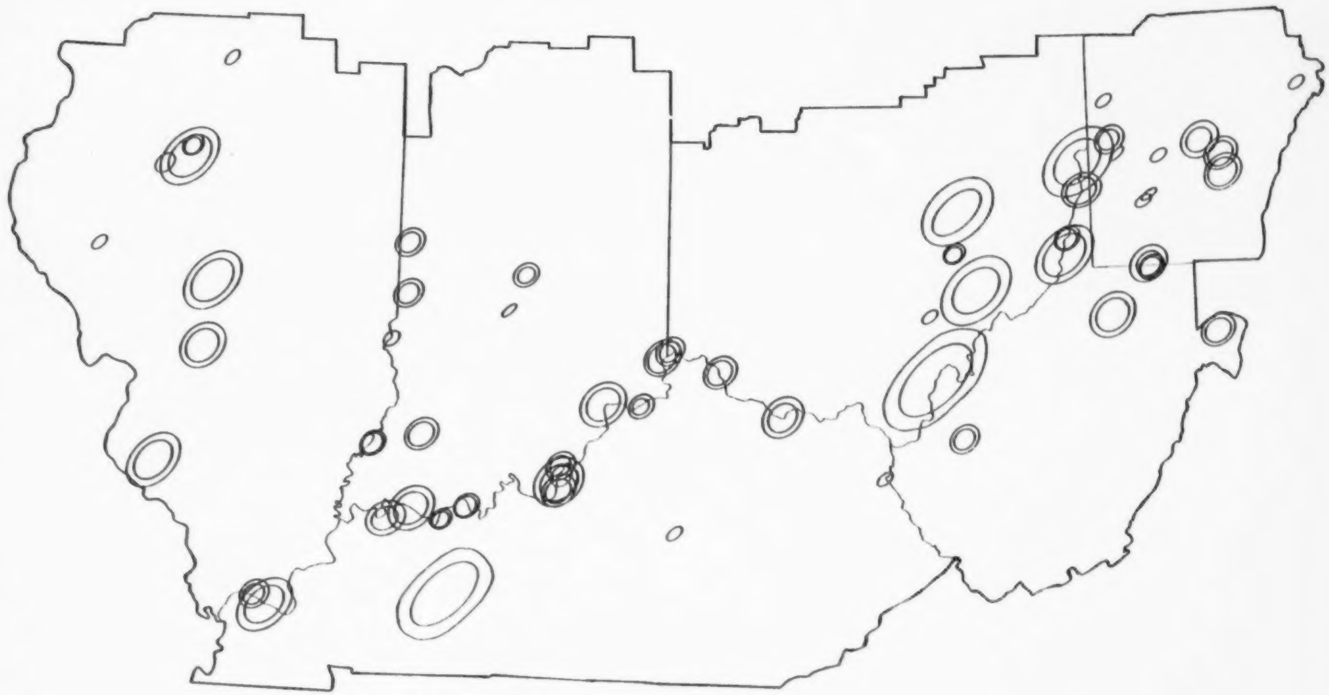

Figure VI-19. 1976. Calculated impact areas of all power plants in the ORBES region with SO_2 emission rates of greater than 2000 g/sec. Outermost ellipse represents isopleth of 0.05 ppm SO_2. Other ellipses, if present, represent 0.1 ppm SO_2 isopleths. See Table VI-4 for listing of power plants, emission rates and impact areas.

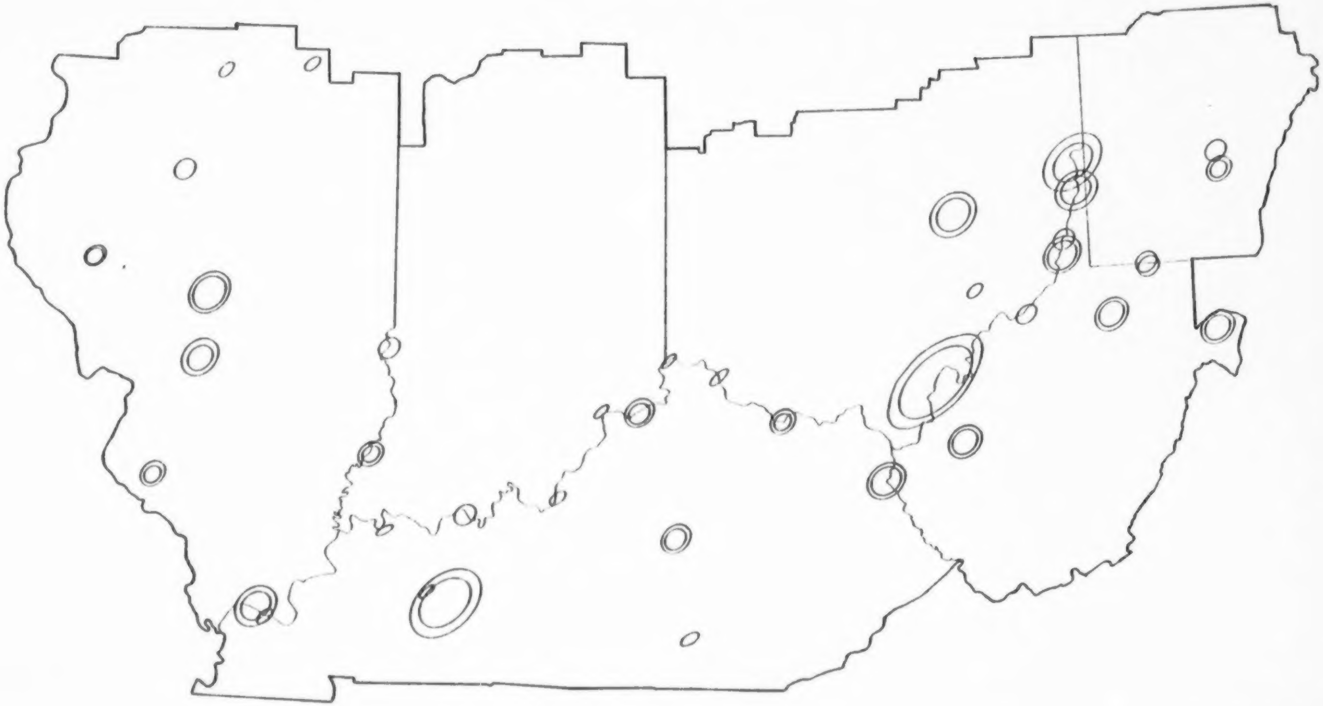

Figure VI-20. 1985. Scenario 2, assuming compliance with SIPs by 1985. Calculated impact areas of all power plants in the ORBES region with SO_2 emission rates of greater than 2000 g/sec. Outermost ellipse represents isopleth of 0.05 ppm SO_2. Other ellipses, if present, represent 0.1 ppm SO_2 isopleths.

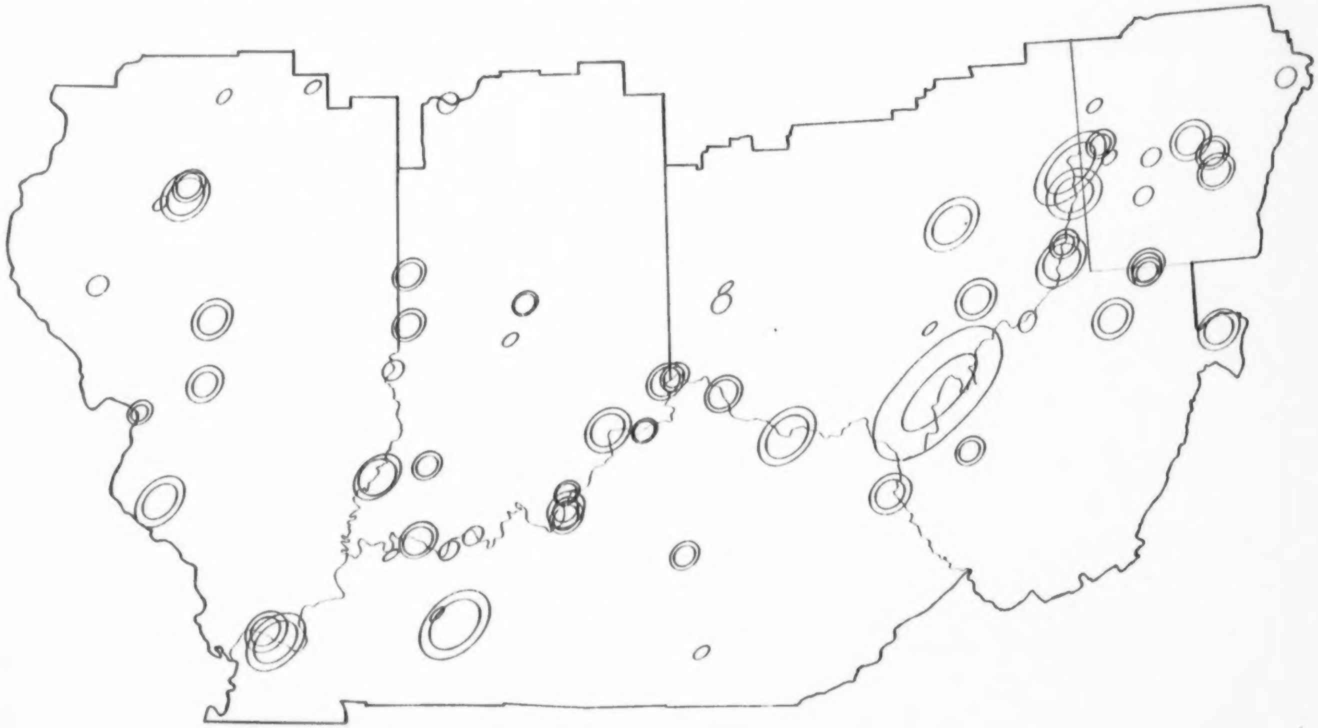

Figure VI-21. 1985. Scenario 2d, assuming no compliance with SIPs. Calculated impact areas of all power plants in the ORBES region with SO_2 emission rates of greater than 2000 g/sec. Outermost ellipse represents isopleth of 0.05 ppm SO_2. Other ellipses, if present, represent 0.1 ppm SO_2 isopleths.

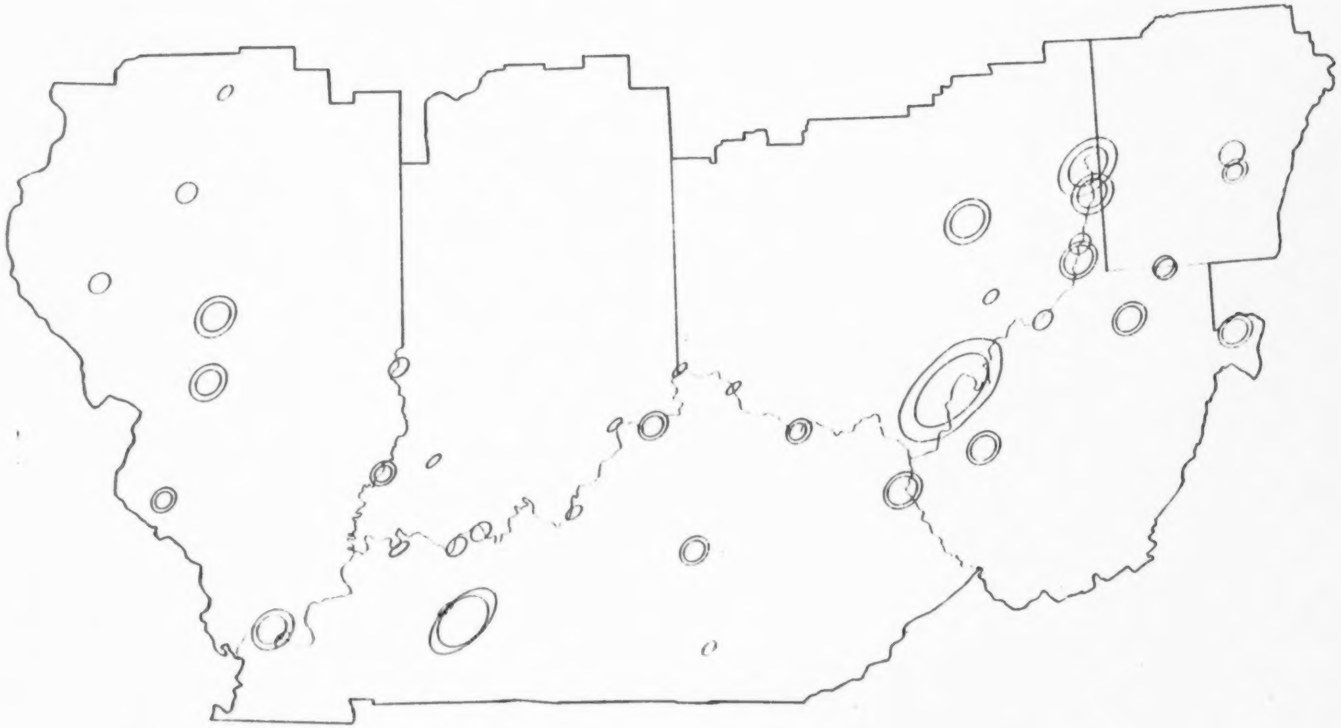

Figure VI-22. 1985. Scenario 7, assuming compliance with SIPs. Calculated impact areas of all power plants in the ORBES region with SO_2 emission rates of greater than 2000 g/sec. Outermost ellipse represents isopleth of 0.05 ppm SO_2. Other ellipses, if present, represent 0.1 ppm SO_2 isopleths.

introduced by using the two sources of data (AIRTEST and USM) in calculating emissions is probably insignificant on a regional basis, but may affect the dimensions of the impact area of an individual power plant.

The total areas within the 0.05 ppm isopleth in each state, and in the entire region, are summarized in Table VI-5. Areas of overlap between nearby sites were estimated and subtracted to avoid double counting. The results show major decreases in the areas with elevated SO_2 concentrations under all three scenarios by the year 2000. Only in 1985 when few refinements of present plants will have taken place, and under lax compliance schedules, are significant increases in projected impact areas expected. Comparison of these data with the maps (Figures VI-19 to 25) indicates the relatively large changes brought about by reducing or eliminating a small number of high-emission plants in several states.

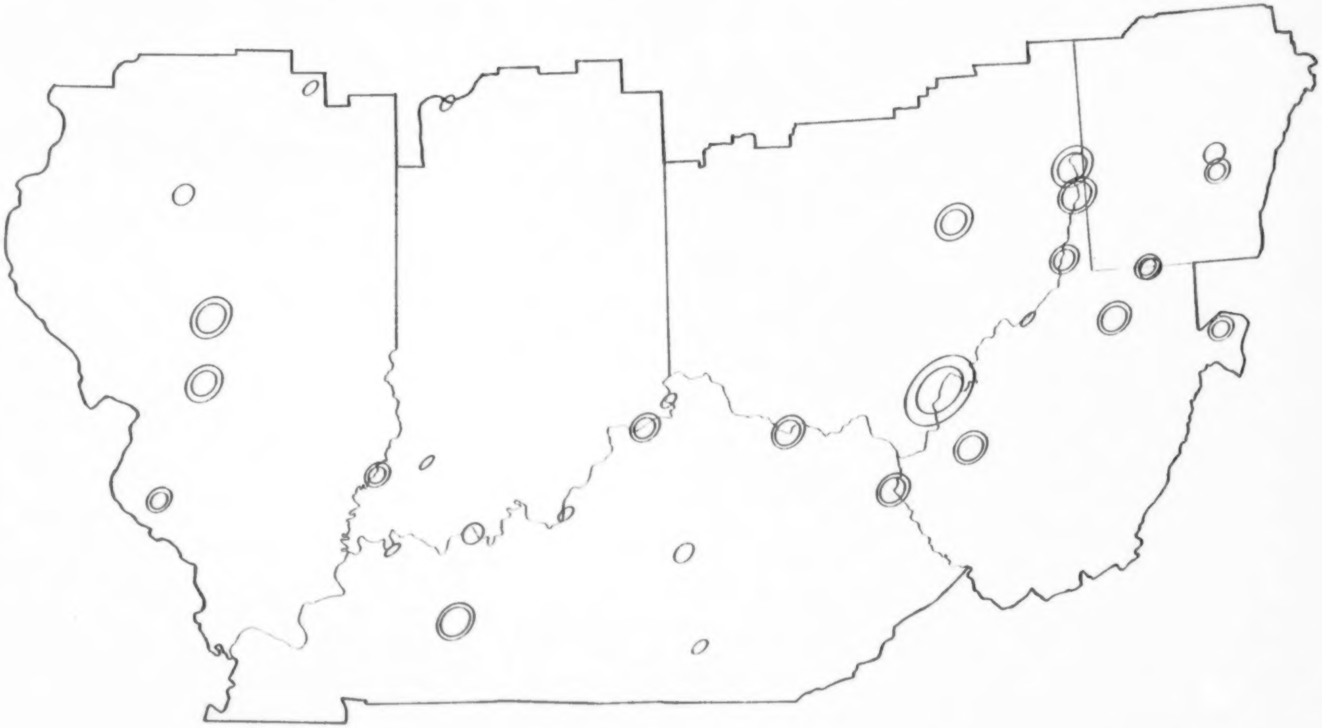

Figure VI-23. 2000. Scenario 2, assuming compliance with SIPs by 1985. Calculated impact areas of all power plants in the ORBES region with SO_2 emission rates of greater than 2000 g/sec. Outermost ellipse represents isopleth of 0.05 ppm SO_2. Other ellipses, if present, represent 0.1 ppm SO_2 isopleths.

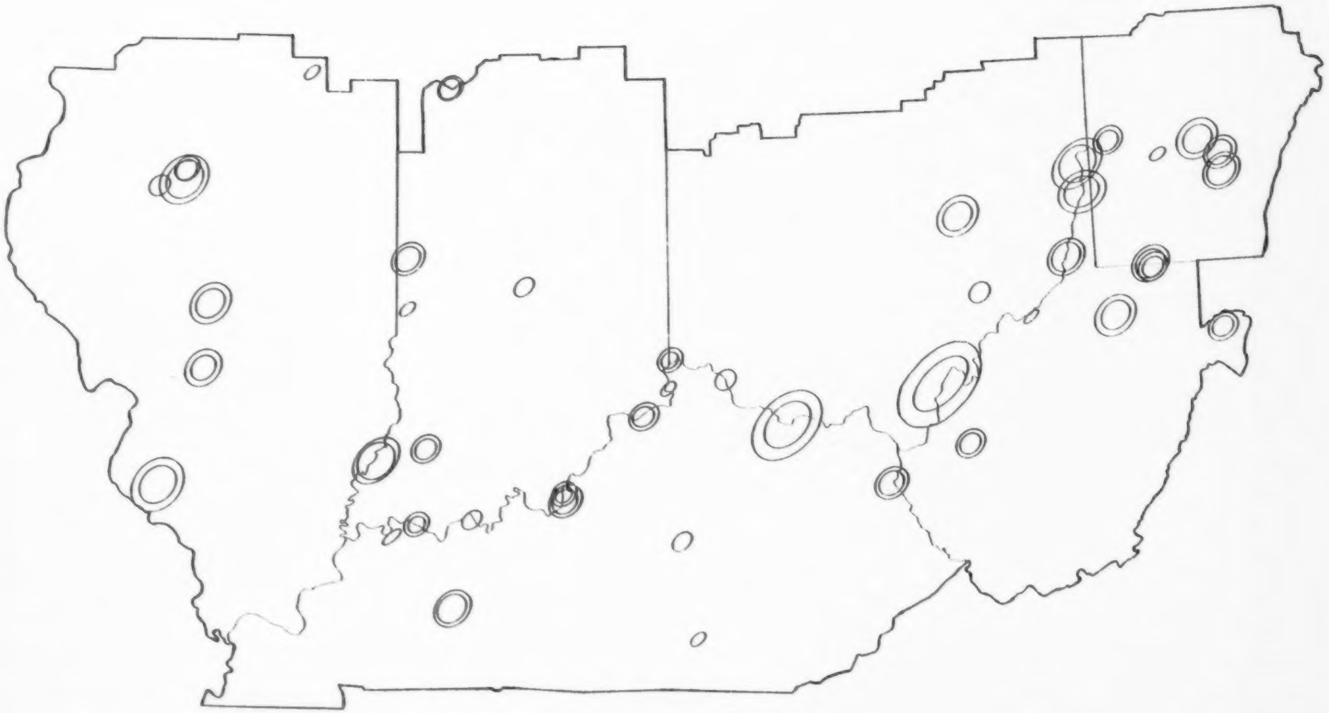

Figure VI-24. 2000. Scenario 2d, assuming no compliance with SIPs. Calculated impact areas of all power plants in the ORBES region with SO_2 emission rates of greater than 2000 g/sec. Outermost ellipse represents isopleth of 0.05 ppm SO_2. Other ellipses, if present, represent 0.1 ppm SO_2 isopleths.

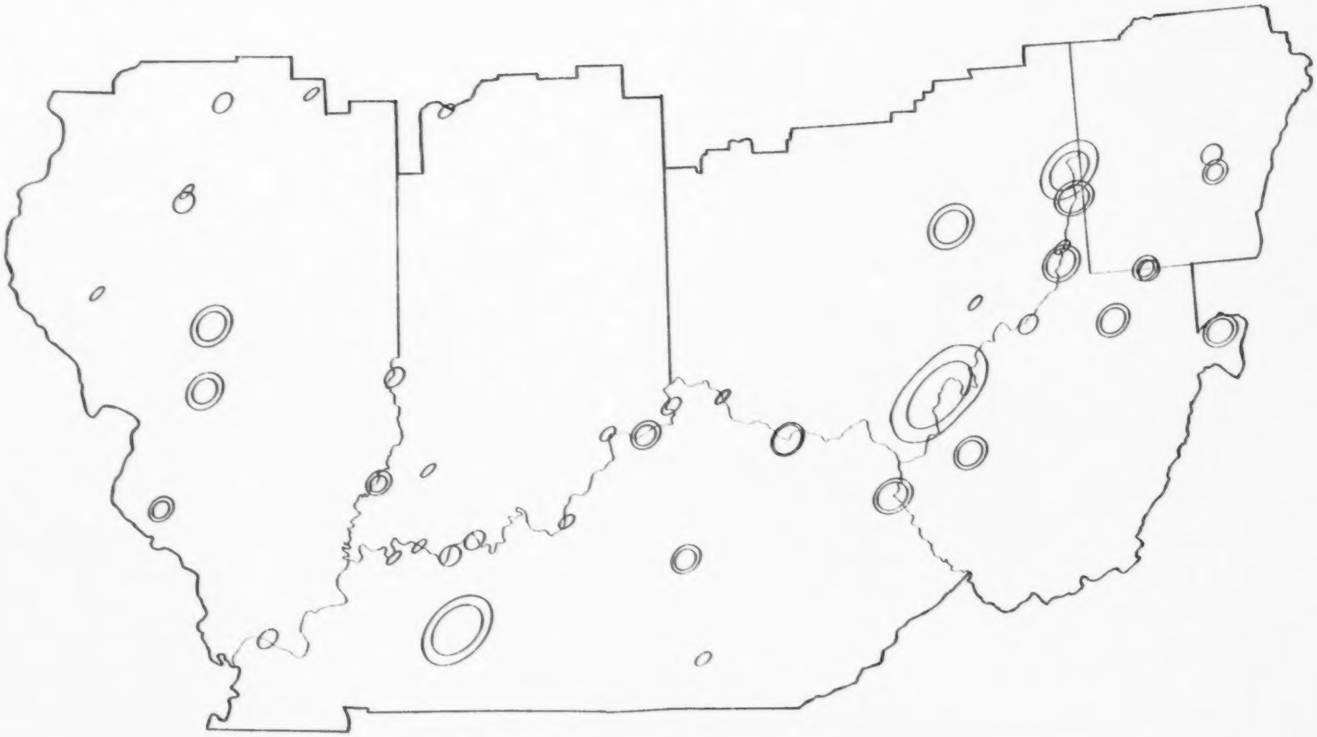

Figure VI-25. 2000. Scenario 7, assuming compliance with SIPs. Calculated impact areas of all power plants in the ORBES region with SO_2 emission rates of greater than 2000 g/sec. Outermost ellipse represents isopleth of 0.05 ppm SO_2. Other ellipses, if present, represent 0.1 ppm SO_2 isopleths.

TABLE VI-5 . SUMMARY OF THE AREA WITHIN THE CALCULATED CONCENTRATION ISOPLETHS EXPRESSED AS A PER CENT OF THE TOTAL AREA OF THE STATE WITHIN THE ORBES REGION

	Total Area Within 0.05 ppm SO_2 Calculated Concentration Isopleth (%)						
State	1976	1985 (2d)[a]	2000 (2d)[a]	1985 (2)[b]	2000 (2)[b]	1985 (7)[c]	2000 (7)[c]
Illinois	6.1	5.0	4.1	3.4	2.2	3.2	2.5
Indiana	7.2	9.1	4.3	0.8	0.7	0.9	1.2
Kentucky	8.8	7.7	3.3	5.1	2.7	5.2	5.3
Ohio	16.6	19.2	11.5	10.4	6.4	10.3	9.5
Pennsylvania	11.1	10.7	10.5	2.0	2.7	1.9	2.7
West Virginia	8.4	8.4	6.2	6.2	4.4	6.2	5.7
Total ORBES	9.3	9.6	6.0	4.7	3.1	4.7	4.5

[a] (2d)- Scenario assuming no compliance with SIPs.

[b] (2) - Scenario assuming compliance with SIPs by 1985

[c] (7) - Scenario assuming high electrical growth, compliance with SIPs by 1985, and extended power plant service life.

DISCUSSION OF SO_2 DISTRIBUTION MODEL

Estimating the effects of air pollutants on vegetation requires the development and application of reliable techniques for calculating the geographic area covered by the pollutant at a specified concentration. For this study, calculations were needed for peak 1-hour SO_2 concentrations that could be expected to affect vegetation during the summer in the vicinity of coal-fired power plants. One approach to the task would have been the routine application of EPA standardized dispersion models, such as CRSTR and RAM. This approach is problematical, since the Pasquill-Gifford dispersion coefficients have not been adequately validated for elevated point sources. Studies have found that the models tend to overestimate concentrations near the plant and underestimate at greater distances (Ragland *et al*. 1979). The development of a more detailed model, or the modification of an existing program, was beyond the scope of this study. Furthermore, the highest short-term concentrations during the summer months would be expected to occur at midday under light wind conditions, a situation which would be less amenable to description with a steady-state Gaussian plume model.

A considerable amount of pollutant monitoring data exists for the ORBES region. This study used data on hourly SO_2 concentrations from five power plants to determine average second-highest concentrations per month, which were plotted as isopleths on monitoring network maps. The elliptical shape of the concentration isopleths, with a northeasterly long axis, appears to be due to the regional meteorology and a high frequency of persistent winds from the southwest (Teknekron 1979).[1] When those days having the highest or second highest 1-hour summertime concentrations were examined, elevated 1-hour concentrations occurred in groups centered about the midday and afternoon hours. During these periods, concentration peaks are generally observed at more than one monitoring site. The synoptic feature that characterizes these periods is the presence of a large high-pressure system, often centered over the upper portion of the Ohio Valley. The presence of these subsidence inversions effectively limits the rise of the plume to less than the height it might otherwise attain. As the atmosphere beneath the inversion becomes more unstable in response to midday heating, portions of the slow-moving stack effluent may be brought back to the surface, a circulation pattern that can persist for several hours.

This rise and fall of SO_2 concentrations during the daylight hours does not seem to be directly related to the phenomenon known as inversion breakup or fumigation, although the latter may have been responsible for several observed 1-hour episodes at the Gavin plant. Inversion breakup involves the gradual elimination of a surface-based inversion until the plume, which was emitted into a very stable layer overnight, is brought to the ground. The associated fumigation process will generally produce elevated concentrations for an hour or less, which is less than the duration of the episodes being observed during the afternoon in the Ohio Valley.

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[1] The contribution of Dr. Michael Mills of Teknekron, Inc, in analyzing meteorological conditions associated with SO_2 trapping events is gratefully acknowledged.

Further analysis of temperature soundings is needed to determine whether the trapping condition is primarily responsible for the observed 1-hour episodes or whether these high concentrations are due simply to an extreme looping condition under light winds. This more intensive examination of 1-hour summer SO_2 episodes should also include an analysis of the following monitoring network parameters for the episode periods in question: ambient temperature, wind speed (10 and 60 meter heights), wind direction, and temperature difference (10 to 60 meter heights). Other relevant parameters, such as weather type, cloud cover, and ceiling height, can be obtained from the nearest National Weather Service Station. Power plant operating parameters for these days can be retrieved from the AEP network data base.

The use of elliptical isopleths to determine impact zones leads to the assumption of a Gaussian distribution of pollutants in the crosswind direction and a Poisson distribution in the downwind direction. The procedure for determining the equation for each curve at each power plant requires cross-sectional data taken from the isopleths. However, the fit of observed data to the calculated lines (Figures VI-6 and 15) indicates limited potential for inaccuracies in this procedure. One of the major problems is the location, and in some cases the small number of monitors. In particular, there are few monitors upwind of power plants, which may tend to skew the isopleths downwind. The most likely error this would cause would be an underestimate of the area of the isopleths on the upwind side. This deficiency cannot be remedied in the current study but should be considered in any application of the method elsewhere.

Another potential problem is the lack of exact fit of the observed data to the calculated Gaussian and Poisson Curves (Figures VI-13 and 14). In particular, the peak predicted concentration for Gavin-Kyger-Sporn (461 $\mu g/m^3$, Figure VI-14) is less than the highest measured concentration (511 $\mu g/m^3$, Figure VI-6). A similar situation occurs at Clifty Creek (Figures VI-14 and 7), and at Breed the reverse is true (Figures VI-14 and 8). The discrepancies are less than 10 percent, could be due to local topographic effects, and do not seem to be biased in one direction or the other. Similarly, comparisons between calculated isopleth areas (Table VI-4) and those measured from actual pollutant distributions for the five monitored power plants (Figures VI-6 to 10) indicate an error of less than 10 percent. All of these considerations support a conclusion that the calculated concentration isopleths are probably not in error by more than 10 percent and that there is no consistent bias.

CONCLUSIONS

Methods have been developed and presented to calculate the areal SO_2 distribution around a power plant, given monitoring at some sites, and the emission rate of SO_2. The model is based on the assumptions that the SO_2 concentrations of interest are those high values that occur several times each month during the growing season, and that these will occur during subsidence inversions when pollutants are trapped in an unstable air mass under a more stable air mass. These conditions are most likely to occur during the growing season in the Ohio River Basin when the wind is moving slowly from the southwest on the "backside" of a high pressure system.

These results lead directly to an assumption of elliptical SO_2 concentration isopleths, with the centers of the ellipses displaced downwind (northeast), so that the distribution of SO_2 is Gaussian in the crosswind direction and Poisson in the downwind direction. The result is a data-based empirical model in which, once the appropriate mathematical functions have been developed, the impact of a single power plant or of all the plants in the region can be determined. The potential SO_2 isopleths of proposed power plants may be estimated prior to operation, which can aid in deciding questions of siting and emission controls.

No significant correlation is found between stack height and impact area. Increased stack height will disperse pollutants over a large area, which probably happens most of the time, but emissions are usually higher as well. Under certain meteorological conditions pollutants may be trapped in the lower atmosphere and accumulate to high concentrations. Since vegetation may be damaged by occasional exposure to high pollutant concentrations, tall stacks with higher emissions do not prevent plant injury. Since some problems are exacerbated by high stacks, e.g., acid precipitation downwind, this strategy for pollutant dispersal may need to be re-examined.

Although the impact area defined using the second highest monthly concentrations is not much affected by stack height, it is dramatically affected by emission rate, rising in an exponential fashion. The practical consequence is that the total impact area in each state, and in the entire ORBES region, can change rapidly as a few large plants are controlled or closed.

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SECTION VI - APPENDIX A

Two computer programs are used in the calculations described in this Section. The purpose of this Appendix is to describe and document those programs. Both are written in BASIC.

The first of the two programs (SO2FIT) sets constants in the equations for Gaussian and Poisson distributions of SO_2 concentrations utilizing specified input data. These equations are:

$D \quad = \quad Cx^{\alpha}e^{-\beta x}$ for the Poisson, and

$D \quad = \quad \alpha e^{-\beta x^2}$ for the Gaussian.

In both of these equations, D is the SO_2 concentration found at distance x from the origin. The origin is upwind of the smokestack in the Poisson distribution, and is the peak (highest D) in the Gaussian distribution.

For the Poisson distribution equation, the constants are C, α, and β. They are found by using the least squares method explained by Hornbeck in Numerical Methods (1975). The equation:

$$\begin{Bmatrix} n & \Sigma \ln x & -\Sigma x \\ \Sigma \ln x & \Sigma(\ln x)^2 & -\Sigma x \ln x \\ -\Sigma x & -\Sigma x \ln x & \Sigma x^2 \end{Bmatrix} \times \begin{Bmatrix} C \\ \alpha \\ \beta \end{Bmatrix} = \begin{Bmatrix} \Sigma \ln D \\ \Sigma \ln x \ln D \\ -\Sigma x \ln D \end{Bmatrix}$$

is used, where n is the number of pairs (x and D) of specified data. A similar method is used for finding the constants α and β in the Gaussian distribution equation.

The following variables are defined in the first program:

X(n) = x where x is the nth distance specified
D(n) = D where D is the nth concentration specified
E(1,1) = lnC (only in the Poisson section of the program)
E(2,1) = α (only in the Poisson section of the program)
E(3,1) = β (only in the Poisson section of the program)
E(1,1) = α (only in the Gaussian section of the program)
E(2,1) = β (only in the Gaussian section of the program)

The second program (CROPLOSS) calculates the area within isopleths of specified SO_2 density, and calculates the estimated crop losses using standard damage coefficients.

The following variables are defined in the second program.

Z or Z(2,9) = A matrix containing the standard damage coefficients.

I = Crop (1 = soybeans, 2 = corn, 3 = wheat)
P(I) = % of county planted with crop I
Y(I) = Normal yield for crop I
E = Power plant emissions (g/sec)
A = Area contained in zone I
B = Area contained in zone II
A(I) = Probable loss for crop I
B(I) = Maximum loss for crop I
C(I) = Minimum loss for crop I

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SO2FIT (Poisson)

```
100  Q=R=T=U=V=W=Y=Z=0
200 DIM M(3,3),D(25),X(25),B(3,3),C(3,1),E(3,1)
300 PRINTCHR$(30);"NUMBER OF PAIRS"
400 INPUT N
500 PRINTCHR$(30);"KEY IN THE PAIRS AS X-VALUE,D-VALUE"
600 FOR I=1 TO N
700 INPUT X(I),D(I)
800 NEXT I
900 FOR I=1 TO N
1000 Q=Q+X(I)
1100 R=R+X(I)*X(I)
1200 T=T+LOG(X(I))
1300 U=U+LOG(X(I))*LOG(X(I))
1400 V=V+X(I)*LOG(X(I))
1500 W=W+LOG(D(I))
1600 Y=Y+X(I)*LOG(D(I))
1700 Z=Z+LOG(X(I))*LOG(D(I))
1800 NEXT I
1900 M(1,1)=N
2000 M(1,2)=T
2100 M(1,3)=-Q
2200 M(2,1)=T
2300 M(2,2)=U

2400 M(2,3)=-V
2500 M(3,1)=-Q
2600 M(3,2)=-V
2700 M(3,3)=R
2800 MAT B=INV(M)
2900 C(1,1)=W
3000 C(2,1)=Z
3100 C(3,1)=-Y
3200 MAT E=B*C
3300 PRINTCHR$(30);"C","ALPHA","BETA"
3400 PRINTCHR$(30);EXP(E(1,1)),E(2,1),E(3,1)
3500 PRINTCHR$(30)
3600 PRINTCHR$(30);"X-VALUE","OBS-D","PRE-D","DIFF"
3700 FOR I=1 TO N
3800 D1 = EXP(  E(1,1)+E(2,1)*LOG(X(I))-E(3,1)*X(I) )
3900 PRINTCHR$(30);X(I),D(I),D1,ABS(D(I)-D1)
4000 NEXT I
4100 PRINTCHR$(30)
4200 PRINTCHR$(30);"KEY IN THE X-VALUES IN ONE LINE SEPARTED BY COMMAS"
4300 MAT INPUT X
4400 PRINTCHR$(30)
4500 PRINTCHR$(30);"X-VALUE","D-VALUE"
4600 FOR I=1 TO NUM

4650 D1 = EXP(  E(1,1)+E(2,1)*LOG(X(I))-E(3,1)*X(I) )
4700 PRINTCHR$(30);X(I),D1
4800 NEXT I
9999 END
```

SO2FIT (Gaussian)

```
100  Q=R=T=U=V=W=Y=Z=0
200 DIM M(2,2),D(25),X(25),B(2,2),C(2,1),E(2,1)
300 PRINTCHR$(30);"NUMBER OF PAIRS"
400 INPUT N
500 PRINTCHR$(30);"KEY IN THE PAIRS AS Y-VALUE,D-VALUE"
600 FOR I=1 TO N
700 INPUT X(I),D(I)
800 NEXT I
900 FOR I=1 TO N
1000 Q=Q+X(I)**4
1100 R=R+X(I)*X(I)
1500 W=W+LOG(D(I))
1600 Y=Y+(X(I)**2)*LOG(D(I))
1800 NEXT I
1900 M(1,1)=-N
2000 M(1,2)=R
2200 M(2,1)=-R
2300 M(2,2)=Q
2800 MAT B=INV(M)
2900 C(1,1)=-W
3000 C(2,1)=-Y
3200 MAT E=B*C
3300 PRINTCHR$(30);"ALPHA","BETA"

3400 PRINTCHR$(30);EXP(E(1,1)),E(2,1)
3500 PRINTCHR$(30)
3600 PRINTCHR$(30);"Y-VALUE","OBS-D","PRE-D","DIFF"
3700 FOR I=1 TO N
3800 D1 = EXP(E(1,1))*EXP(-E(2,1)*(X(I)**2))
3900 PRINTCHR$(30);X(I),D(I),D1,ABS(D(I)-D1)
4000 NEXT I
4100 PRINTCHR$(30)
4200 PRINTCHR$(30);"KEY IN THE Y-VALUES IN ONE LINE SEPARTED BY COMMAS"
4300 MAT INPUT X
4400 PRINTCHR$(30)
4500 PRINTCHR$(30);"Y-VALUE","D-VALUE"
4600 FOR I=1 TO NUM
4650 D1 = EXP(E(1,1))*EXP(-E(2,1)*(X(I)**2))
4700 PRINTCHR$(30);X(I),D1
4800 NEXT I
9999 END
```

CROPLOSS

```
100 FILE #1:"DA,MILLERTIE,SYS052 "
200 DIM Z(2,9)
300 PRINT CHR$(30);
400 PRINT "NEW CROPS (Y/N)";
500 INPUT S$
600 IF S$="N" THEN 2400
700 PRINT CHR$(30);
800 PRINT "CROP TYPE 1"
900 FOR I=1 TO 2
1000 INPUT Z(I,1),Z(I,2),Z(I,3)
1010 PRINT CHR$(30);Z(I,1),Z(I,2),Z(I,3)
1100 NEXT I
1200 PRINT CHR$(30);
1300 PRINT "CROP TYPE 2"
1400 FOR I=1 TO 2
1500 INPUT Z(I,4),Z(I,5),Z(I,6)
1510 PRINT CHR$(30);Z(I,4),Z(I,5),Z(I,6)
1600 NEXT I
1700 PRINT CHR$(30);
1800 PRINT "CROP TYPE 3"
1900 FOR I=1 TO 2
2000 INPUT Z(I,7),Z(I,8),Z(I,9)
2010 PRINT CHR$(30);Z(I,7),Z(I,8),Z(I,9)

2100 NEXT I
2200  RESET #1:0
2300 MAT WRITE #1:Z
2350 GOTO 2600
2400 RESET #1:0
2500 MAT READ #1:Z
2525 T$=CHR$(30)&"        CROP 1                  CROP 2                  CROP 3"
2530 PRINT T$
2540 S$=CHR$(30)
2550 S$=S$&"-#.#   -.    -.    -.    -.    -.    -.    -.   "
2575 MAT PRINT USING S$,Z
2600 PRINT CHR$(30);
3000 READ M1,B1,M3,B3
3100 DATA .0012,2.1,.00078,2.21
3200 READ M2,B2,M4,B4
3300 DATA 2.,-87.7,0.8,-22.6
3400 DIM P(3),Y(3),A(3),B(3),C(3)
3450 PRINT CHR$(30);
3500 PRINT "ENTER PLANT NAME"
3600 INPUT P$
3650 PRINT CHR$(30);
3700 PRINT P$
3800 FOR I=1 TO 3

3900 S$="ENTER %(.XX), YIELD FOR CROP -#"
3950 PRINT CHR$(30);
4000 PRINT USING S$,I
4100 INPUT P(I),Y(I)
4150 PRINT CHR$(30);
4175 PRINT P(I),Y(I)
4185 PRINT CHR$(30)
4200 NEXT I
4250 PRINT CHR$(30);
4300 PRINT "ENTER EMISSION"
```

```
4400 INPUT E
4450 PRINT CHR$(30);
4500 PRINT E
4600 FOR I=1 TO 3
4700 R1=M1*E+B1
4800 R2=SQR(M2*SQR(E)+B2)
4900 A=R1*R2*2010.62
5000 R1=M3*E+B3
5100 R2=SQR(M4*SQR(E)+B4)
5200 B=R1*R2*2010.62
5300 A1=A-B
5400 C1=P(I)*A1
5500 C2=P(I)*B

5600 L=3*(I-1)
5700 C3=Y(I)/(1-Z(1,3+L))
5800 C4=Y(I)/(1-Z(2,3+L))
5900 A(I)=C1*C3*Z(1,3+L)+C2*C4*Z(2,3+L)
6000 B(I)=C1*C3*Z(1,2+L)+C2*C4*Z(2,2+L)
6100 C(I)=C1*C3*Z(1,1+L)+C2*C4*Z(2,1+L)
6200 NEXT I
6225 PRINT CHR$(30)
6250 PRINT CHR$(30);
6275 PRINT "ZONE I AREA   ZONE II AREA"
6280 PRINT CHR$(30);
6285 PRINT A,B
6286 PRINT CHR$(30)
6287 PRINT CHR$(30);
6290 PRINT "          SOYBEANS"
6295 GOSUB 10000
6300 PRINT A(1),B(1),C(1)
6303 PRINT CHR$(30)
6305 PRINT CHR$(30);
6308 PRINT "          CORN"
6310 GOSUB 10000
6320 PRINT A(2),B(2),C(2)
6321 PRINT CHR$(30)

6322 PRINT CHR$(30);
6325 PRINT "         WHEAT"
6330 GOSUB 10000
6340 PRINT A(3),B(3),C(3)
6380 PRINT CHR$(30)
6400 PRINT "ANOTHER EMISSION? (Y/N)"
6500 INPUT S$
6600 IF S$="Y" THEN 4300
6700 PRINT"ANOTHER PLANT (Y/N)"
6800 INPUT S$
6900 IF S$="Y" THEN 3500
7000 STOP
10000 PRINT CHR$(30);
10100 PRINT "PROBABLE      MAXIMUM        MINIMUM"
10200 PRINT CHR$(30);
10300 RETURN
10400 END
```

SECTION VII

CALCULATION OF TOTAL PRODUCTION LOSS OF SELECTED CROPS DUE TO DIRECT IMPACT OF SULFUR DIOXIDE

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INTRODUCTION

The objective of this section is to apply the yield loss functions developed in Section V to the impact areas calculated in Section VI to determine the effects on agricultural yields and total crop production of present and future levels of sulfur dioxide (SO_2) emissions.

The two previous sections have described how the area of direct SO_2 emission impact in the ORBES region can be determined using an empirical model based on SO_2 measurements, and how the effects of SO_2 on crop yield have been incorporated into loss coefficients for peak exposures of 0.1 and 0.05 parts per million (ppm) SO_2 twice per month. Both of these calculations are necessary for the computation of total crop production effects in the entire ORBES region. The goal in this section is to estimate the aggregate reduction in crops, which, upon mitigation, can be considered a benefit in any emissions abatement program. Although reference is made to crop losses associated with the downwind impacts of coal-fired power plants, recovery from these losses ultimately must be viewed as a potential benefit to be weighed against the benefits of more abundant or cheaper electricity.

Other estimates of agricultural losses due to air pollution have been summarized by Leung et al. (1978), Noorbakhsh et al. (1979), and Heagle et al. (1979). Some estimates were based on experts' crop surveys for visible damage, which is related to losses of yield in poorly understood ways. Others have utilized various regression techniques, usually for large geographic areas, and often with questionable assumptions and data. Most studies have been concerned with losses due to oxidants. In almost all of these studies there was no attempt to define an area of direct impact, or to relate loss of yield to the actual crops grown in a region. One study that attempted to relate vegetation damage directly to SO_2 from a specific power plant was reported by Linzon et al. (1979). Impact zones around the power plant were mapped, using observed vegetation damage symptomatic of SO_2. Meteorological factors and chemical studies of the plants were taken into account. The study did not, however, attempt to estimate loss of yield or economic implications. Jaksch (1979) has reviewed many past attempts at evaluating air pollution damage to crops and pointed out their shortcomings.

The effects of a pollutant on a plant in the field may be very different from effects determined under controlled conditions in the laboratory. Changing agricultural practices, such as increased fertilization and the use of pollution-resistant varieties, may alter the effects of pollutants. This section uses reported crop yield and acreage data to determine losses due to SO_2 in defined impact areas around power plants. Regional impact of SO_2 is calculated by summing the effects of individual power plants. Crop losses from projected SO_2 emissions in the future have been calculated on the basis of ORBES projections of power plant construction and operation to the year 2000. Loss functions at different SO_2 concentrations, as described in Section V, should be more realistic than laboratory-derived values. However, this section does not attempt to account for changing agricultural practices in the future. The results are reported as estimated reduction in annual yield, converted to total site, state and regional losses in production, expressed as bushels. No attempt is made here to estimate economic loss in dollars.

Direct measurement of crop yield reductions due to SO_2 has not been attempted because it would require a major research program beyond the scope of this study. Simple comparisons of yield inside and outside impact zones are confounded by variables such as soil type, rainfall, agricultural practices, etc. Simple field comparisons also would be unlikely to quantify SO_2 damage around power plants in the ORBES region because of the high impact of elevated oxidant levels (see Section V). Vegetation damage within the SO_2 impact areas is demonstrated in Section IV for white pines, which, because of their over-wintering foliage, are more likely to show visible injury than the crop species. There appears to be a small gradient within Indiana of increasing air pollution injury from rural sites without power plants, to rural areas within SO_2 impact zones, to urban areas. The gradient is gradual, however, emphasizing the difficulty of doing field studies within a region characterized by multiple stresses.

METHODS

The procedures for applying the crop loss coefficient to the projected clean air yield free of O_3 effects (see Section V) and multiplying by crop acreage is illustrated in Table VII-1. Crops were selected based on economic importance, potential effect, and availability of data for determining loss functions.

Computations were made for one or two impact zones (column B), dependent on emissions levels around each plant using the elliptical isopleth method described in Section VI. Impact Zone I, for example, includes all of the area between the 0.05 ppm SO_2 isopleth and the 0.10 ppm SO_2 isopleth. Impact Zone I, therefore, is the area around the power plant where peak SO_2 concentrations are between 0.05 and 0.10 ppm at least twice in each summer month. Impact Zone II can be similarly described.

The total crop area impacted (column C) was calculated as a product of the proportion of the county acreage (see State Agricultural Statistics

TABLE VII-1. EXAMPLE CALCULATION OF ESTIMATED PRODUCTION LOSS CAUSED BY SO_2 IN THREE CROPS AROUND A SINGLE COAL-FIRED POWER PLANT

Scenario 1976

Crop	A % of Impacted County (3 yr. avg.)	B Impacted Area (Acres)	A x B = C Crop Area Impacted (Acres)	D Loss Function (%) Min	 Max	 Prob[c]	E Probable Clean Air Yield of Impacted County[d] (3 yr. avg.)	C x D x E = F Loss of Production (Bushels) Min	 Max	 Prob
Soybeans	26.98	I[a] 147,920	I 39,910	I 1	4	3	41.0 Bu./Ac.	I 16,365	64,450	49,090
		II[b] 125,285	II 33,800	II 3	9	6	42.3 Bu./Ac.	II 42,890	128,675	85,785
Corn	44.85	I 147,920	I 66,340	I 0	2	0	120.1 Bu./Ac.	I 0	159,350	0
		II 125,285	II 56,190	II 0	3	2	122.5 Bu./Ac.	II 0	206,500	137,665
Wheat	2.26	I 147,920	I 3,343	I 1	3	2	43.8 Bu./Ac.	I 1,464	4,393	2,928
		II 125,285	II 2,831	II 3	6	5	45.2 Bu./Ac.	II 3,839	7,679	6,399

[a] I = 0.05-ppm SO_2 impact zone (see text)

[b] II = 0.10-ppm SO_2 impact zone (see text)

[c] See Section V of this report

[d] Actual Yield: Soybeans = 39.77 Bu./Ac.; Corn = 120.1 Bu./Ac.; Wheat = 42.9 Bu./Ac.

Annual Reports cited) covered by each crop and the impact area for each SO_2 concentration. This assumes that crops are distributed within the impact zone in the same way they are distributed within the entire county. This could introduce a small error for a few power plants along the Ohio River whose impact zones may have topography and therefore crops different from those in the rest of the county.

Three different loss functions (expressed as percentages) were used: the lower bound or minimum estimate, the upper bound, or maximum, and the most probable (see Section V). Because the objective of this section is to determine crop losses due to SO_2 pollution, and because the three-year actual yields of each county already reflect projected losses due to sulfur dioxide, a probable clean air yield had to be calculated (column E). Thus, the crop loss estimates are calculated from the estimated crop yield expected if no air pollution were present, i.e., equivalent to the SO_2-free air used for "control" yields in some of the controlled-environment studies (Section V). The formula for calculating the clean air yield is:

$$\text{Probable Clean Air Yield} = \left(\frac{\text{3-year average yield}}{\text{100 - Probable } SO_2 \text{ loss function}}\right) \times 100$$

For example, to determine the probable clean air yield for soybeans in Zone I (Table VII-1), divide 39.77 by 97 (100% - 3%), and multiply by 100, to obtain 41 bushels per acre as the "probable" clean air yield. Thus, current annual reduction in soybean yield because of power plant SO_2 emissions is estimated to be 1.2 bushels per acre in Zone I, and 2.5 bushels per acre in Zone II. The product of the loss function, probable clean air yield for the county, and impacted crop area gives the estimated loss in production for each impact zone (column F). This loss of production is equivalent to the projected gain in production which would be expected from abatement of the SO_2 emissions.

RESULTS AND DISCUSSION

From these estimated gains around each plant, the total potential production gain was summed for each state and for the entire ORBES region. The total probable gain for 1976 and for the three scenarios considered in this report are shown in Tables VII-2 to 4. Table VII-5 gives the range of crop gains occurring in the entire ORBES region for all years and scenarios. Current crop yield data has been used for future projections, although possible changes in agricultural practices (e.g., growing of more pollution-resistant species) could influence the results considerably, as a considerable cost in conversion.

No attempt was made to estimate fully the effects of overlapping impact zones from somewhat remote upwind power plants (see Section VI). Obviously an area within the 0.05 ppm isopleth (Zone I) of two power plants will actually receive an exposure equivalent to Zone II. In the case of a crop such as soybeans, where the Zone I loss function is half

TABLE VII-2. ESTIMATED INCREASES OF SOYBEAN PRODUCTION DUE TO ABATEMENT OF PEAK-LOAD SO_2 EMISSIONS FROM COAL-FIRED ELECTRICAL GENERATING STATIONS IN THE ORBES REGION. (Gain expressed in thousands of bushels)

State	Probable Clean Air Production 3-year average 1975-1977 (1000's of bu)	1976 Gain	1985 (2)[a] Gain	2000 (2)[a] Gain	1985 (2d)[b] Gain	2000 (2d)[b] Gain	1985 (7)[c] Gain	2000 (7)[c] Gain
Illinois	309,154	905	456	349	741	612	444	377
Indiana	130,935	265	35	36	349	225	37	48
Ohio	76,366	39	9	7	65	34	9	10
Pennsylvania	None	None	None	None	None	None	None	None
Kentucky	38,030	263	78	25	176	30	78	72
West Virginia	None	None	None	None	None	None	None	None
TOTAL ORBES	554,485	1,472	578	417	1,331	901	568	507

[a] (2) -- Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

[b] (2d) -- Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

[c] (7) -- Scenario assuming high electrical growth, compliance with SIPs by 1985, and extended power plant service life

TABLE VII-3. ESTIMATED INCREASES OF CORN PRODUCTION DUE TO ABATEMENT OF PEAK-LOAD SO_2 EMISSIONS FROM COAL-FIRED ELECTRICAL GENERATING STATIONS IN THE ORBES REGION. (Gain expressed in thousands of bushels)

State	Probable Clean Air Production 3-year average 1975-1977 (1000's of bu)	1976 Gain	1985 (2)[a] Gain	2000 (2)[a] Gain	1985 (2d)[b] Gain	2000 (2d)[b] Gain	1985 (7)[c] Gain	2000 (7)[c] Gain
Illinois	1,087,860	712	337	269	579	534	337	269
Indiana	596,688	313	48	48	441	333	48	48
Ohio	287,709	122	64	45	116	81	63	63
Pennsylvania	16,590	22	7	8	24	24	7	8
Kentucky	127,075	220	61	26	120	31	61	61
West Virginia	2,678	6	4	4	6	5	4	4
TOTAL ORBES	2,118,600	1,395	521	400	1,286	1,008	520	453

[a] (2) - Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

[b] (2d) - Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

[c] (7) - Scenario assuming high electrical growth, compliance with SIPs by 1985, and extended power plant service life

TABLE VII-4. ESTIMATED INCREASES OF WHEAT PRODUCTION DUE TO ABATEMENT OF PEAK-LOAD SO_2 EMISSIONS FROM COAL-FIRED ELECTRICAL GENEPATING STATIONS IN THE ORBES REGION. (Gain expressed in thousands of bushels)

State	Probable Clean Air Production 3-year average 1975-1977 (1000's of bu)	1976 Gain	1985 (2)[a] Gain	2000 (2)[a] Gain	1985 (2d)[b] Gain	2000 (2d)[b] Gain	1985 (7)[c] Gain	2000 (7)[c] Gain
Illinois	70,499	203	87	74	188	155	87	76
Indiana	53,717	105	18	17	142	92	20	21
Ohio	45,801	29	15	11	36	22	15	15
Pennsylvania	1,238	5	1	1	5	5	1	1
Kentucky	11,150	31	7	4	21	5	7	7
West Virginia	121	1	1	None	1	1	1	1
TOTAL ORBES	182,526	374	129	107	393	280	131	121

[a] (2) - Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

[b] (2d) - Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

[c] (7) - Scenario assuming high electrical growth, compliance with SIPs by 1985, and extended power plant service life

TABLE VII-5. ESTIMATED PRODUCTION INCREASES OF THREE CROPS DUE TO ABATEMENT OF PEAK LOAD SO_2 EMISSIONS FROM COAL-FIRED ELECTRICAL GENERATING STATIONS. (Gain is for the entire ORBES region, expressed in thousands of bushels.)

Crop	1976	1985 (2)[a]	2000 (2)[a]	1985 (2d)[b]	2000 (2d)[b]	1985 (7)[c]	2000 (7)[c]
Soybeans							
Probable Gain	1,472	578	417	1,331	901	568	507
Maximum Gain	2,124	831	604	1,923	1,304	813	727
Minimum Gain	653	255	187	592	404	252	220
Corn							
Probable Gain	1,395	521	400	1,286	1,008	520	453
Maximum Gain	3,529	1,341	1,007	3,240	2,416	1,321	1,274
Minimum Gain	None	None	None	None	None	None	None
Wheat							
Probable Gain	374	129	107	393	280	131	121
Maximum Gain	479	166	135	503	358	167	156
Minimum Gain	214	74	61	225	160	75	69

[a] (2) - Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

[b] (2d) - Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

[c] (7) - Scenario assuming high electrical growth, compliance with SIPs by 1985, and extended power plant service life

that of Zone II, counting Zone I losses twice gives the same overall result as counting Zone II losses once. In the case of corn and wheat, where Zone I loss functions are less than half those in Zone II, counting Zone I losses twice underestimates the true losses. Considering the amount of overlap, these underestimates will vary between scenarios but are unlikely to exceed a few percent of the total loss for any single state or scenario.

The data in Tables VII-2 to 4 show that for all years and scenarios most of the estimated gain in production due to abatement of SO_2 around ORBES coal-fired electrical generating stations is expected in one state, Illinois. A significant gain also is evident in Indiana. This is because of the very large agricultural acreages around power plants in these two states, while power plants in other states tend to be surrounded by forest.

In addition, a large part of the estimated probable gain of soybeans and corn for the entire ORBES region could be achieved by emission abatement at three Illinois power plants. The large losses incurred around these plants are a function of high SO_2 emission rates, large areas of agriculture, and high yields; the last two are most significant.

Tables VII-2 to 4 also show that the total gain of production due to SO_2 emission abatement is a very small fraction of the normal production of the entire region (less than 1%). From this perspective, the direct effects of SO_2 emissions in the ORBES region might be thought of as almost negligible. However, when gains attributed to individual power plants are expressed in relation to crop yields and profit margins for farm operators in the counties receiving the greatest impact, the effect is more significant, at least for that audience. In Illinois, for example, the estimated gain of soybeans due to the operation of one power plant is more than 3% of the total county soybean production. The gain to individual farmers in proximity to such a power plant would be larger than 3% (see Table VII-1). Other county comparisons range from 3% to less than 1% for soybeans and corn. The significance of these yield differences in relation to profitability for annual farm operation costs should be explored further. These results suggest the importance of distinguishing between regional and more local impacts.

The results presented in this section are subject to the limitations on estimates of impact area and the loss coefficients discussed in previous sections of this report. However, when faced with high levels of uncertainty, every effort has been made to take the conservative estimate and to avoid an accumulation of errors in one direction.

CONCLUSIONS

1. When considered on a regional scale, the total potential gain in crop production from abatement of SO_2 emissions from coal-fired electrical generating stations is almost insignificant at present and is expected to decrease under the compliance scenarios considered. On a

local scale, i.e., the county level, the gain from abatement may be quite significant and the gain to individual farmers near the sources of SO_2 may be substantial. Gains will vary slightly under the various scenarios.

2. Over half of the estimated direct gain in production of soybeans, corn and wheat is expected to occur in Illinois. Furthermore, from 40 to 60% (depending on the scenario used) of the total ORBES gain in soybeans production could be achieved by SO_2 emission controls at three power plants in Illinois. The same values for corn are 30-50% and for wheat, 20-45%.

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SECTION VIII

CROP LOSSES DUE TO OXIDANTS CAUSED BY EMISSIONS FROM COAL-FIRED POWER PLANTS IN THE OHIO RIVER BASIN

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INTRODUCTION

The analysis of crop loss functions (Section V) has shown ozone (O_3) to be the gaseous pollutant of greatest interest in the ORBES area. However, no precise data are available as to the historic time-trend of O_3 concentrations in this area. One could assume that O_3 in the Ohio Basin is presently unchanged from historic levels, and that anthropogenic emissions are not contributing to any detectable change. Although assuming zero change in O_3 is a possibility, there is little evidence supporting it. Thus, the question becomes one of determining from the indirect evidence available (such as the observable time-trends in effects on vegetation) what the magnitude of the societally induced increases on O_3 may have been, and could continue to be over the span of ORBES power plant development. If increasing O_3 in the rural areas is supported scientifically, a second question for ORBES is whether the NO_x emissions from utilities have had a zero, or non-zero, contribution to the increases.

Ideally, the examination of these questions requires a fully validated model of O_3 formation processes in rural areas where the concentrations of hydrocarbon and NO_x precursors are relatively low. Such models (e.g., Antell 1979) are barely beyond the preliminary testing stages, at present. Thus, the options for this study, and for the ORBES work as it relates to resource effects, were to accept both the "zero change in O_3" and "zero contribution of utilities" assumptions above or examine qualitatively the evidence on apparent trends in O_3, its effects and precursors, and as appropriate, draw conclusions expressed in terms of probable or possible responses.

Thus, the objective of this section is to examine the evidence and the uncertainties in evaluating crop losses in the Ohio River Basin associated with current or future concentrations of O_3. The crop loss functions derived in Section V indicate O_3 toxicity for crops is of much greater significance than SO_2 toxicity. To evaluate the issues presented to TIE researchers in the projections of increased coal emission in the ORBES area (see Section I), the potential for changes in O_3 concentrations in the future had to be considered as a possibility. The following information would be needed to evaluate this potential for O_3 effects on crops:

1. The present concentrations of O_3 in rural areas in the Ohio River Basin;

2. Evidence regarding present natural O_3 background as opposed to apparent anthropogenic additions;

3. Evidence on rate-limiting factors in O_3 formation in largely rural areas, e.g., oxides of nitrogen and reactive hydrocarbons;

4. Data relating to the differentiation of several origins and long-distance transport of O_3 and its precursors, e.g., automobile exhausts, agricultural releases of nitrogen and power plant residuals;

5. Estimates on historical O_3 trends over time in relation to trends in O_3 precursors, including projections to the year 2000.

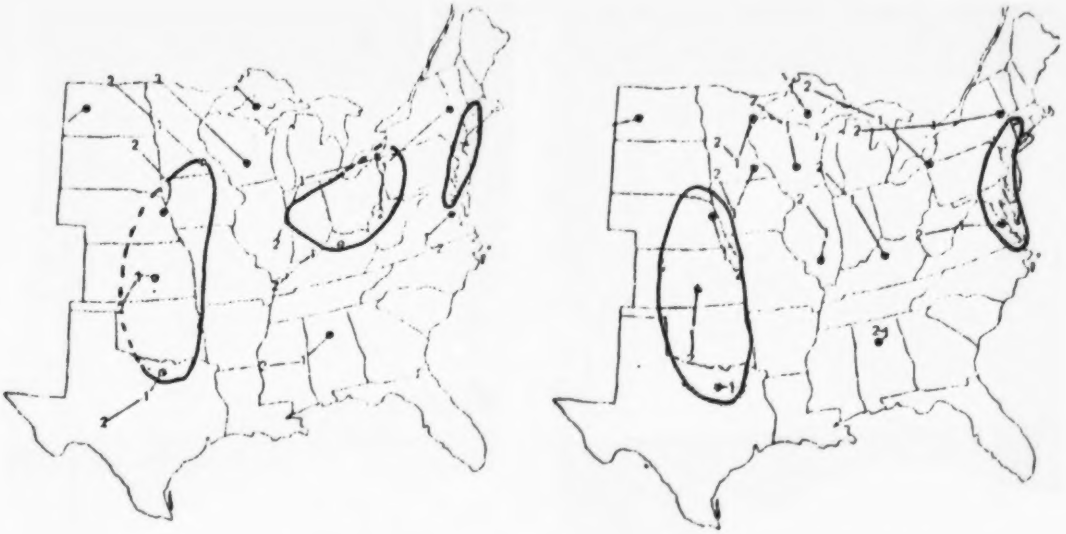
The following sections, while not intended to be monographic, examine a representative part of the recent information base available on each of these issues.

PRESENT CONCENTRATIONS OF O_3 IN THE OHIO RIVER BASIN

At monitoring sites throughout the ORBES region, concentrations of O_3 have been observed frequently at levels capable of inducing negative responses from vegetation and crops (see Section V; also see National Academy of Sciences 1976, Haering 1977, Cleveland and Graedel 1979, Lioy and Sampson 1979, Heagle et al. 1979, Illinois EPA 1979). Concentrations of O_3 in the range of 0.07 to 0.20 ppm occur with some regularity in rural areas throughout the Ohio River Basin.

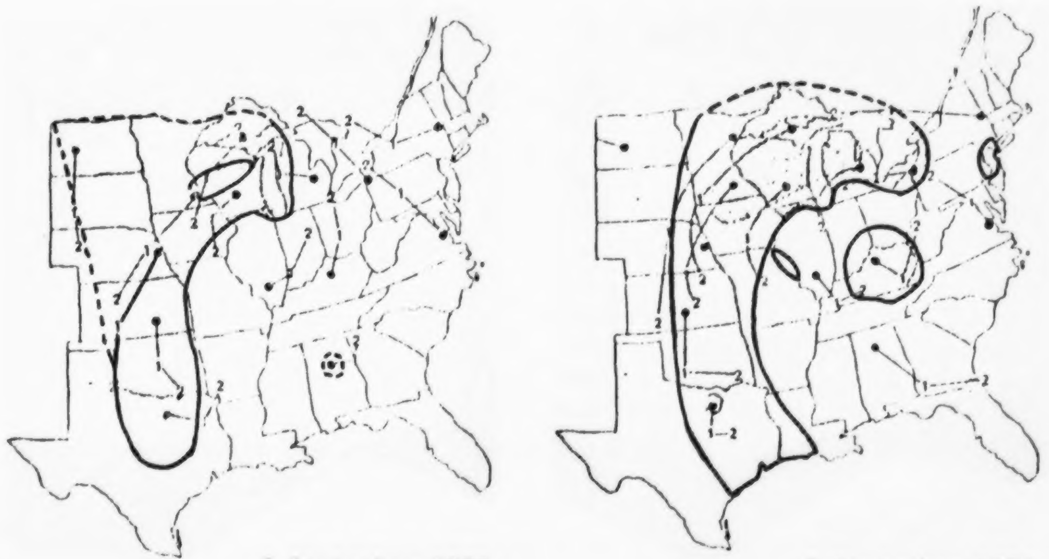
The duration of O_3 exposures in the Ohio River Basin also is a factor in considering the magnitude of potential crop reductions in this region. The peak O_3 values are observed during periods of gradual build-up of O_3, often after the passage of a slow-moving high pressure system. Studies by Bowen (1978) and by Wolff and Lioy (1980) have traced the movement of representative stagnated air masses with elevated O_3. Bowen showed that beginning September 4, 1976, in the Texas-Oklahoma area, an air mass with increasing elevated O_3 moved north to the upper Great Lakes states and on to New England by September 10 (Figure VIII-1). A similar analysis by Wolff and Lioy (1980) reached the conclusion that O_3 is formed rather continuously in these air masses as the O_3 is broken down and scavenged by a variety of mechanisms.

The processes reflected in these papers result in a deep blanket of heavily polluted air, extending with rather uniform concentrations of O_3 to 8,000 or 9,000 ft. (Figure VIII-2). These relatively well-mixed high-particulate and high-oxidant layers move across the region between major weather events. The overnight depletion of O_3 in the lower 500 feet of these systems (Figure VIII-3) suggests that uptake by vegetation or other ground-level surfaces causes the diurnal cycle in rural areas. Replenish-



4 September 1976

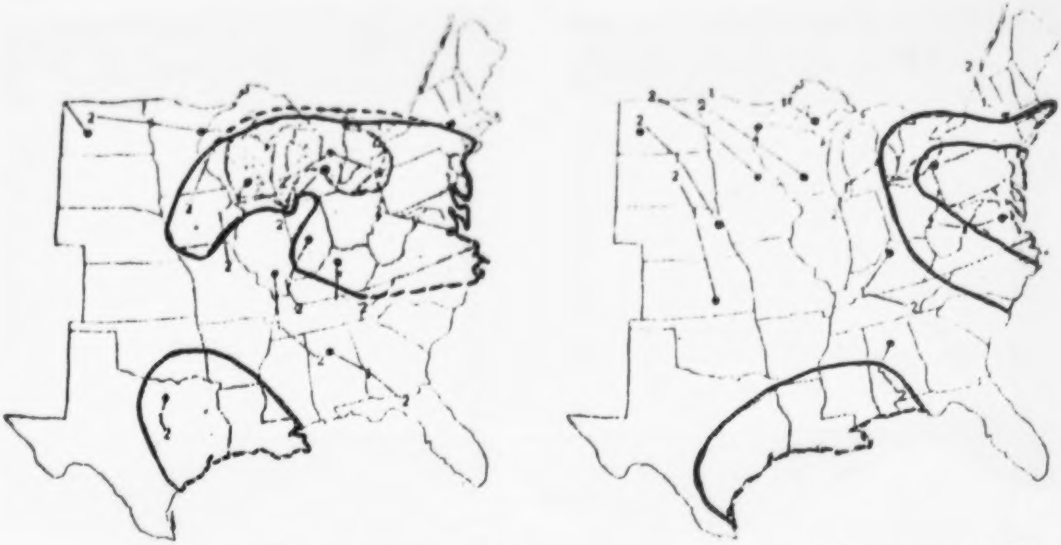
5 September 1976



6 September 1976

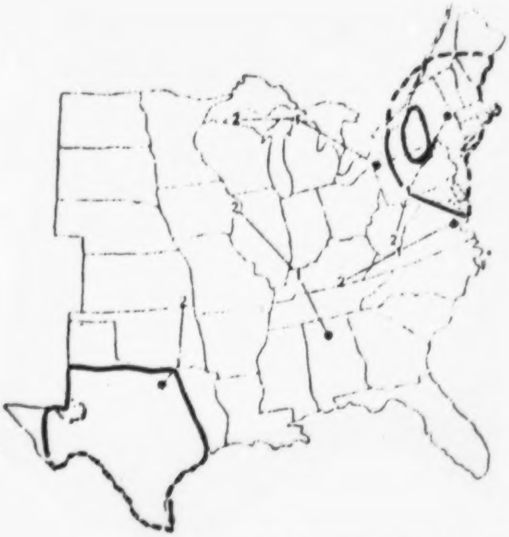
7 September 1976

Figure VIII-1. Surface analysis of the movement of air masses across the eastern United States containing O_3 exceeding 70 ppb from 4 September through 7 September 1976. Source: Bowen 1978.



8 September 1976

9 September 1976



10 September 1976

Figure VIII-1. (continued) Surface analysis of the movement of air masses across the eastern United States containing O_3 in excess of 70 ppb from 8 September through 10 September 1976. Source: Bowen 1978.

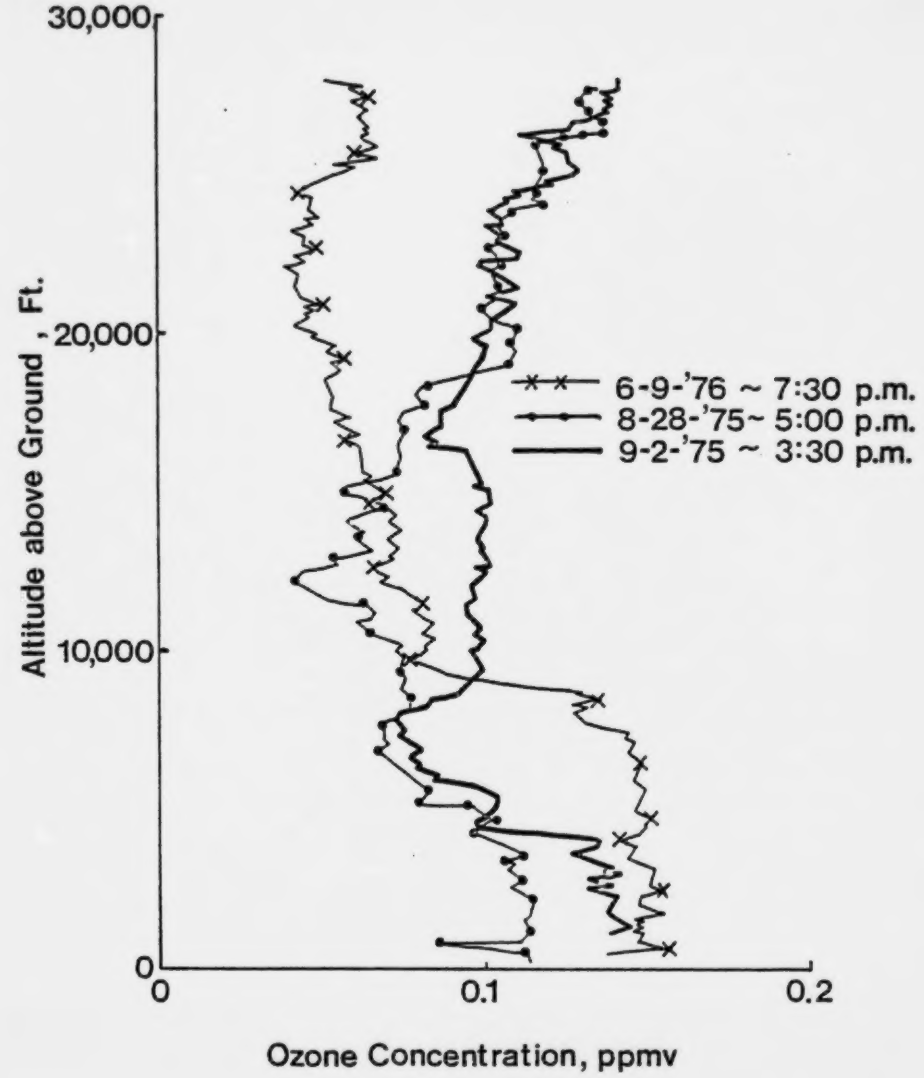


Figure VIII-2. Ozone concentration profile in the area of Indianapolis, Indiana, at three dates in 1975-76. Source: Haering 1977.

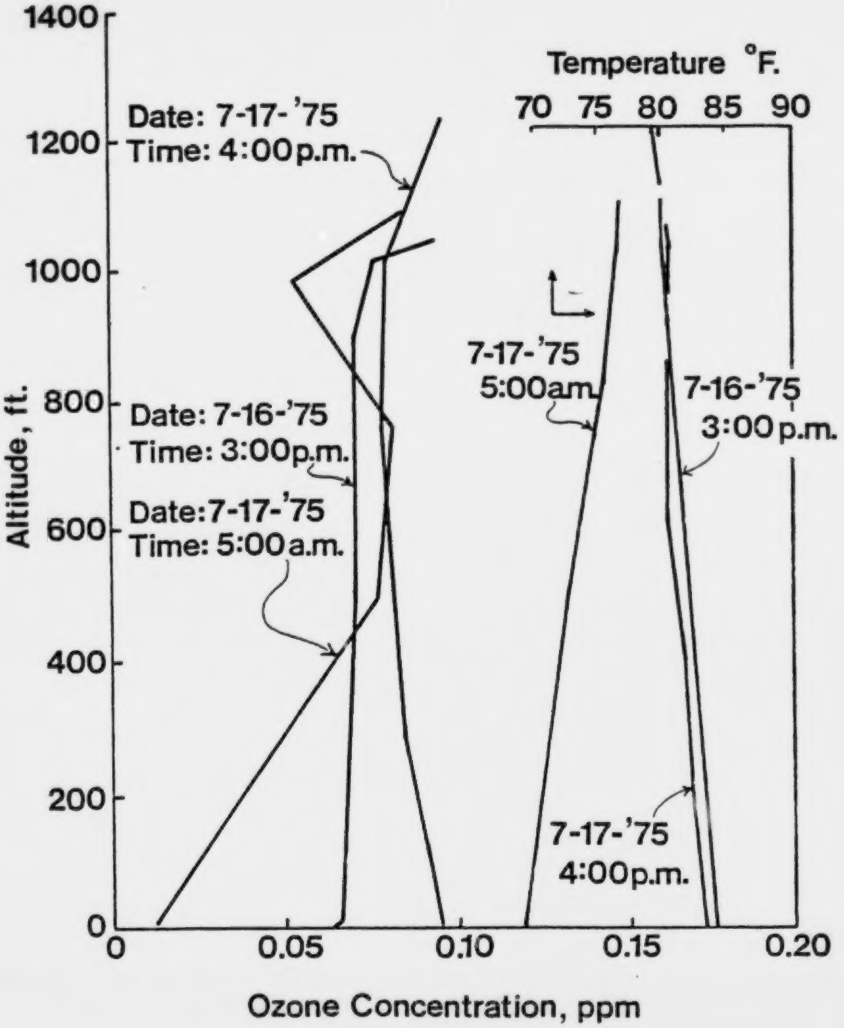


Figure VIII-3. Ozone concentration profile at Indianapolis, Indiana, at two dates in 1975-76. Source: Haering 1977.

ment of the O_3 at the ground layer can be from the 1,000-9,000 ft. layer as heating and turnover of the lower atmosphere take place during daylight hours, although newly formed O_3 in urban areas may also contribute (Haering 1977).

THE SOURCES OF O_3

Natural Background

One major issue is the question of how much of the ambient O_3 observed in the Ohio Basin is of natural origin. There is general agreement that over the ocean, ambient levels of O_3 rarely rise above 0.03 to 0.05 ppm (Haagen-Smit and Wayne 1976). The exceptions are brief episodes during or after thunderstorms or strong cyclonic activity, when the stratospheric O_3 (usually above 35,000 ft.) may break through and increases O_3 concentrations in the lower atmosphere.

Although the stratosphere contains 5 to 10 ppm O_3, the lower stratosphere is resistant to vertical motion, and hence to large volume exchanges of substances with the trophosphere below (Biswas 1979, National Academy of Sciences 1976). Thus, short-term drawdowns of O_3 from the stratosphere cannot be considered the principal cause of sustained O_3 exposures at low elevations. Lightning-induced O_3 also has been measured locally, but, due to its short duration is not suspected of causing regional crop losses. Thus, there is little evidence on which to conclude that natural sources of O_3 contribute to the vegetation stress observed on eastern white pines (Section IV) and other plant species in the Ohio River Basin.

The Anthropogenic Sources of O_3

Ozone formation downwind from major sources of nitrogen oxides (NO_x) and non-methane hydrocarbons is well-documented (NAS 1976, Antell 1979, Spicer et al. 1979). There has been doubt, however, as to whether such reactions would occur downwind from point-source coal-fired power plants (EPRI 1976). Many of these point sources already are located within urban air masses, and single plume effects usually cannot be distinguished. The increase in O_3 in the urban plume itself along a gradient downwind of St. Louis was documented by White et al. (1979). Peak O_3 levels were observed at 50 km downwind, and the urban influence was still detectable above background at 100 km.

The recent report of Miller et al. (1978) shows that O_3 does form in coal-fired power plant plumes when conditions are suitable. Measurements of SO_2, particulates, and O_3 concentrations were made in the plume of the Oak Creek coal-fired plant in Wisconsin as it moved across Lake Michigan (Figure VIII-4a-c). At a cross-section about 20 km downwind, a peak in SO_2 and particulates was observable, as well as the expected depression in O_3 due to scavenging by plume effluents such as nitrogen oxides, from about 90 parts per billion (ppb) to about 70 ppb. Thus, NO_x emissions from the stack scavenged only up to 25% of the ambient O_3 at this location. Halfway across Lake Michigan, the SO_2 spike was still detectable, with a flattening and a broadening of the particulate distribution across

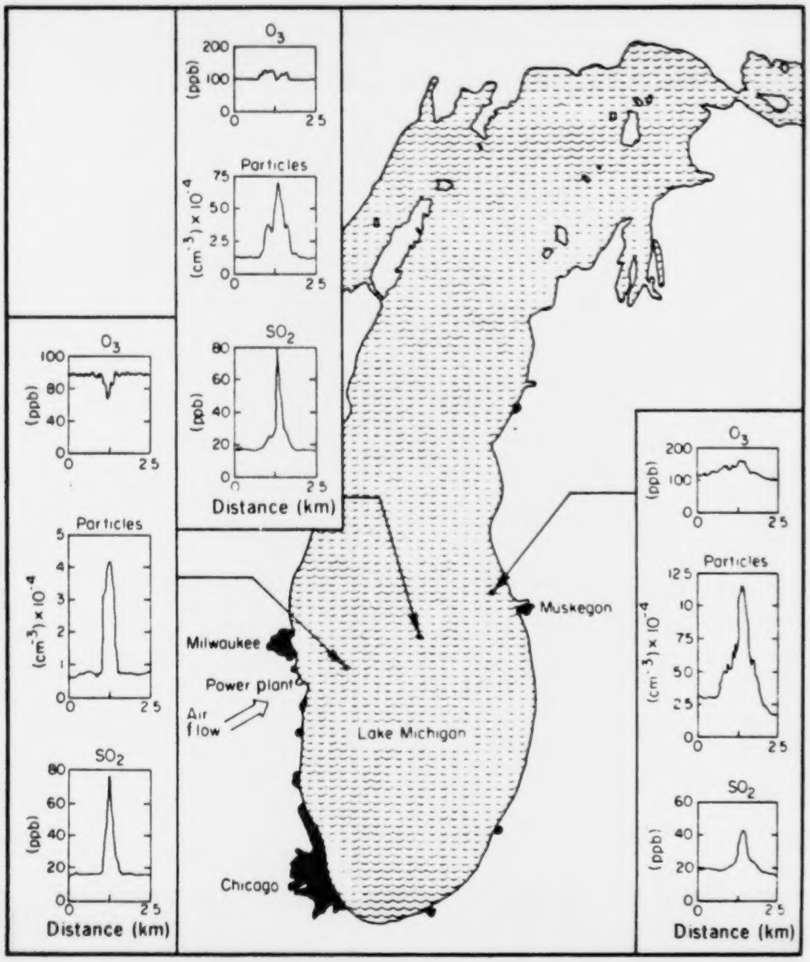
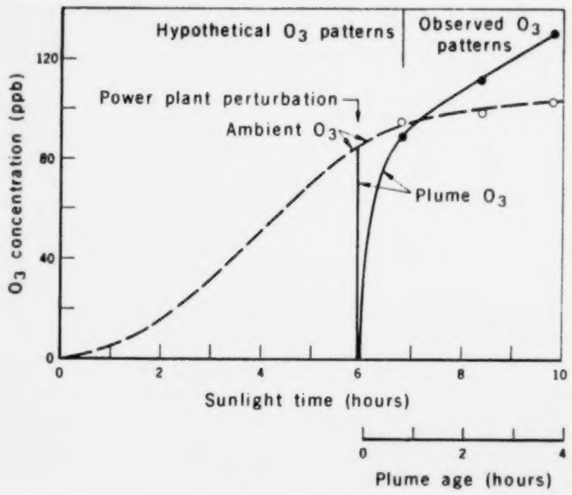

O₃
(ppb)
200
100
0
0
25
Particles
(cm⁻³) x 10⁻⁴
7.5
5.0
2.5
SO₂
80
60
40
20
Distance (km)
100
80
60
40
5
4
3
2
1
Muskegon
12.5
10
Milwaukee
Power plant
Air flow
Lake Michigan
Chicago

Figure VIII-4a. Aerometric data showing the concentration of SO_2 particles larger than 0.01 µm, and O_3 in a power plant plume traveling 300 m above Lake Michigan. Source: Miller *et al*. 1978.



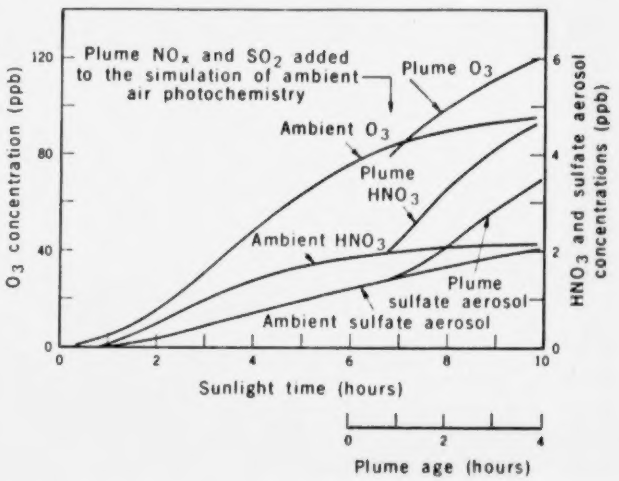


Figure VIII-4b. (left) Trends in the average O_3 concentrations as observed in the air downwind of a power plant.

Figure VIII-4c. (right) Theoretical predictions of the excess O_3, HNO_3, and sulfate aerosol in the air downwind of a power plant.
Source: Miller _et al_. 1978.

the plume. The O_3 concentration had increased to in the order of 120 ppb. By the time the plume had reached the east side of the lake, much of the SO_2 had diffused, but particulates had increased, in part, a result of sulfate aerosol formation from the SO_2. The O_3 concentration on the east side of the lake had increased to about 150 ppb, documenting the so-called "ozone bulge" that had been difficult to observe over land.

The results of the Miller et al. (1978) work indicate that under these weather conditions power plant emissions can produce a 20 to 30% increase in O_3. The authors note they may have observed unique weather conditions, since stronger winds would favor atmospheric mixing and prevent NO_x and hydrocarbons from undergoing rapid reaction to form O_3. They suggest the atmosphere would usually be too turbulent to allow the primary precursors to exist at sufficient concentrations for reactions forming O_3. However, plumes often are observed remaining intact over long distances on land as well as over water, having been reported by Pettyjohn and McKeon (1976) using satellite photography in the Ohio River Basin (Figure VIII-5). In the Ohio River Basin, trapping events and stagnation episodes are frequent (Bowen 1978, Haering 1977), allowing build-up of significant concentrations of NO_x, SO_2, and hydrocarbons similar to those in the Oak Creek plume. Although additional studies are needed, the most plausible interpretation at this time seems to be that various types of weather conditions can produce air parcels favoring O_3 formation from the emissions of power plants in rural areas. Obviously, urban sources of NO_x and hydrocarbons also contribute in considerable measure to the O_3 concentrations being observed in the agricultural areas of the ORBES region (Antell 1979).

NITROGEN OXIDES AND HYDROCARBONS AS RATE LIMITING FACTORS IN RURAL AREAS

Although the chemical processes involved in O_3 formation are the subject of considerable research, the broad outlines are known (see Stern 1976, Haagen-Smit and Wayne 1976, Isaken et al. 1978, Cleveland and Graedel 1979). When light energy is available and temperatures appropriate, non-methane hydrocarbons and nitrogen oxides react to form a variety of oxidant compounds, of which O_3 is predominant. Most of the studies relating to O_3 formation from NO_x and hydrocarbon precursors have been carried out in urban areas where concentrations of precursors can be moderately high and reaction times are short (Calvert 1976). However, Isaksen et al. (1978) and Antell (1979) draw attention to the high O_3 concentrations frequently observed over large rural areas. These measurements and the associated models of Antell demonstrate that O_3 apparently can be formed from relatively low concentrations of precursors in rural areas outside of any identifiable urban plume.

The Sources and Quantities of Hydrocarbons

The greatest part of the non-methane hydrocarbons (NMHC) contributing to O_3 formation in urban areas has generally been attributed to automobile emissions. In comparatively arid regions, vegetation has a short growing season with much less productivity than in the more humid regions of the east. Thus, in the rural areas of the east, both natural and anthropo-

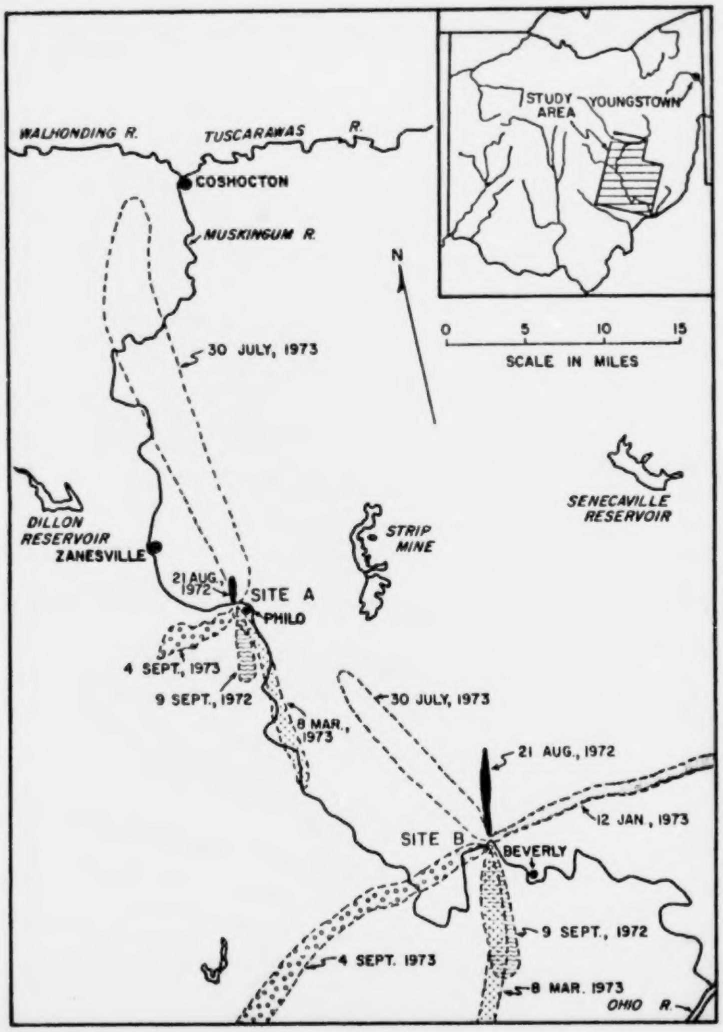

WALHONDING R.
TUSCARAWAS R.
COSHOCTON
MUSKINGUM R.
STUDY AREA
YOUNGSTOWN
N
0 5 10 15
SCALE IN MILES
30 JULY, 1973
SENECAVILLE RESERVOIR
STRIP MINE
DILLON RESERVOIR
ZANESVILLE
21 AUG. 1972
SITE A
PHILO
4 SEPT., 1973
9 SEPT., 1972
8 MAR., 1973
30 JULY, 1973
21 AUG., 1972
12 JAN, 1973
SITE B
BEVERLY
9 SEPT., 1972
4 SEPT. 1973
8 MAR. 1973
OHIO R.

Figure VIII-5. Map based on LANDSAT-1 data showing areal extent of smoke plumes originating at Philo and Beverly, Ohio. Source: Pettyjohn and McKeon 1976.

genic sources of hydrocarbons need to be considered with a view to determining whether, in the Ohio Valley, they function frequently as a rate-limiting factor in O_3 formation.

Plant components of the biosphere release a complex array of nonmethane hydrocarbons with RO_2, RCO_3, and HO_2 radicals. Several of the volatile hydrocarbons that originate from vegetation are judged probably to be reactive in O_3-forming processes (Bufalini _et al_. 1976, Tingey and Burns 1980). Rasmussen (1970) observed that isoprenes emanate from oak leaves, aspen, and sweetgum foliages in the field. Other species capable of significant isoprene volatilization include weeping willow, eastern cottonwood, red oak, sweetgum, and sycamore--all common in the Ohio Valley area. Conifer foliage, on the other hand, releases monoterpenes, instead of isoprene, some of which may be species-specific.

Tingey and Burns (1980) recently reviewed data on hydrocarbon emission rates from vegetation and, depending on location, found biogenic hydrocarbon emissions can range from 12% greater to 20% less than anthropogenic emissions. Calculations based on the natural source emissions estimates of Tingey and Burns (1980) and the anthroprogenic sources of Wolff and Lioy (1980), suggests that, for deciduous forests, emissions are 6.6 $mg \cdot m^{-2}hr^{-1}$ during daylight hours, and the anthropogenic hydrocarbon emission densities are 0.8 $mg \cdot m^{-2}hr^{-1}$ for rural areas and 4.0 $mg \cdot m^{-2}hr^{-1}$ for large urban centers. Thus, from both sources, a substantial background of reactive hydrocarbons would be expected during the growing season in rural areas of the Ohio Valley, sufficient for ozonation reactions if NO_x is present at the necessary concentrations. Indeed, although utility contributions to the hydrocarbons are relatively small, the manageable (i.e., societal) contribution of the reactive NMHC appears to be smaller than the manageable portion of NO_x sources.

As with the long distance O_3 transport phenomenon already presented, hydrocarbons of both natural and anthropogenic origins can be accumulated and carried great distances in slow-moving, summer air masses of the eastern United States. Study of the formation of O_3 over a wide range of time scales (Isaksen _et al_. 1978) suggests the time for O_3 formation is a few hours in an urban plume, but over rural areas may be several days.

The Sources of Elevated NO_x Over Rural Areas

Summaries by Drysdale and Kalif (1977) and other ORBES reports[1] indicate that the majority of NO_x emissions in the ORBES area originates from autos, industry in urban areas, and coal-fired power plants. Atmospheric concentrations of NO_x in the rural areas have been monitored in the vicinity of several coal-fired power plants in the Ohio Valley area (Table VIII-1). Peak concentrations of NO_x range from 0.029 to 0.053 ppm.

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[1] "_Ohio River Basin Energy Study (ORBES) Main Report_." Office of Research and Development. U.S. Environmental Protection Agency.

TABLE VIII-1. NO_x CONCENTRATIONS AT FOUR RURAL SITES IN THE VICINITY OF COAL-FIRED POWER PLANTS IN THE OHIO RIVER BASIN

Power Plant	NO_x Concentrations ($\mu g/m^3$) Average of Second Highest Hourly Values by Month (June-Aug. 1975-1977)		Average (ppm)
	Station 1	Station 2	
Gavin/Kyger/Sporn	74.3 (.039 ppm)	55.7 (.029 ppm)	.034
Clifty Creek	101.2 (.053 ppm)	68.0 (.036 ppm)	.045
Breed	60.6 (.032 ppm)	62.9 (.033 ppm)	.033
Amos	75.2 (.040 ppm)	----	.040

The data listed are averages of the second-highest hourly values (a measure used for SO_2) during each month of the growing season from June through August, 1975-1977. Peak individual observations are appreciably higher, and lower concentrations occur during logarithmically longer periods of time.

Thus, during the conditions of stable, slow-moving air masses characteristic of O_3 episodes, relatively large parts of the rural atmosphere downwind from large power generating stations contain NO_x in the .003 ppm range used in simulating rural smog formation (Antell 1979). These concentrations represent ground-level, multiple episodes in a mosaic of time and geographic displacement throughout the region (see maps in Section VI). Since much of the NO_x from power plant emissions would remain in a plane above ground level (see Miller et al. 1978), ground-level power plant emissions data may underestimate the time, space or concentration characterization of NO_x available for ozone reactions in the Ohio River Basin. However, depending on mixing rates, hydrocarbon concentrations at the elevation of stack plumes could be somewhat lower, due to their origin near the ground surface.

Hydrocarbon and NO_x Ratios

Numerous studies have shown that O_3-forming reactions function at an optimal mix of NMHC and NO_x. Depending on the actual composition and reactivity of the hydrocarbons, the optimal ratio may be in the order of 10 to 20, although results by Antell (1979) show an optimal ratio of 100:1. Concentrations of both NMHC and NO_x vary widely in rural areas, but the results of Lonneman et al. (1978), showing NMHC ranging from 0.05 to 0.20 ppm, are probably representative of summertime high-pressure systems. As noted above, significant rural areas of the Ohio Valley areas are exposed to NO_x concentrations between .003 and .05 ppm.

Calvert (1976) and Antell (1979), suggested that HC/NO_x ratios can be viewed as a sliding scale, varying with the movement of air parcels on the landscape. For example, an air parcel moving downwind from a power plant plume or urban area, while still in the vicinity of the source, will contain NO_x that is elevated but background hydrocarbons would remain about the same, reducing the HC/NO_x ratio. As the NO_x is diluted, however, the ratio of HC to NO_x will increase, increasing ozonation. Reactions in the lower range would be somewhat hydrocarbon-limited, and at the upper end of the range would be NO_x-limited.

Since the ground-monitored NO_x concentrations in the Ohio River Basin range from below detection level to 0.050 ppm (Table VIII-1), and NMHC appears to range from below .03 to above .2 ppm, the ratios of reactive hydrocarbons to NO_x extend in both directions from a range of less than 10 to above 100 where NO_x is being diluted over rural areas. This range is sufficient for ozone formation to occur (Haagen-Smit and Wayne 1976, Isaksen et al. 1978, Antell 1979).

Daly (1979) examined O_3 production rates in Sydney, Australia, in relation to various HC/NO_x ratios to evaluate O_3 control strategies there. He concluded that the simultaneous reduction of HC and NO_x lowers the range of potential O_3 throughout a wide range of HC/NO_x ratios. Cleveland and Graedel (1979), however, note that simultaneous reduction of anthropogenic HC and NO_x might increase O_3 in metropolitan areas, but would reduce rural O_3. These researchers (Cleveland and Graedel 1979) and others (Daly 1979) conclude that NO_x is the principal limiting constituent in their study areas and that NO_x reduction is essential for O_3 control. Additional NO_x from the projected power plants, therefore, must be viewed as most probably contributing to increased O_3 in the Ohio River Valley.

These results also suggest that the hydrocarbons and much of the O_3 in western portions of the Ohio River Basin will have originated outside the region. Thus, another question emerges as to whether the effects induced by these constituents are equivalent to effects induced in regions east of the Ohio Valley area by NO_x and O_3 from within the region following passage beyond the eastern side of the basin. The slow passage tends to allow the products of local reactions to accumulate, producing effects both inside and outside of the region. Although much new work needs to be done, the assumption is made for this study that ORBES area induced effects outside the region can be substituted for the equivalent effects in the region produced by non-ORBES pollutants.

THE POTENTIAL FOR UTILITY NO_x EMISSIONS TO CONTRIBUTE TO O_3

The previous sub-sections have presented evidence suggesting that NO_x emissions in the ORBES area, regardless of source, have a potential to contribute to O_3 formation. Thus, utility NO_x emissions seem most probably to be a contributor. This subsection considers the question of what magnitude this contribution may be.

Drysdale and Kalif (1977) computed the amount of NO_x attributable to societal activity for the United States. Table VIII-2 displays this information for West Virginia, Indiana, Illinois, and Ohio. Electricity generation in these Ohio River Basin states is estimated to have produced 1.086 billion kg of NO_x annually during the baseline years of 1973-1975, or 32% of the total annual NO_x pollution in those states.

In estimating prospective upper and lower bounds of O_3 attributable to electricity generation, differences in the elevation of emissions (stack heights) and the elevation of NO_x scavenging processes, especially of low-level sources by vegetation, need be taken into consideration. Emissions from sources other than power plants are generally at or near ground level and, due to the rapid gaseous uptake of NO_x by vegetation, are less likely to enter photochemical conversion processes. If, at lower concentrations, the Ohio River Basin vegetation absorbed 55% of local NO_x (Rogers et al. 1979 reported 65% mean absorptive capacity for NO_x in snapbeans), then the relative proportion of power plant emissions available for oxidant-forming reactions would be 51%.

TABLE VIII-2. SOURCES OF OXIDES OF NITROGEN IN FOUR STATES IN THE OHIO RIVER BASIN FROM 1973-1975.[1]

State	Source	Amount (10^6 kg/yr)	Percentage	Population Change 1960-1970	Population (10^6)	Per Capita NO_x
West Virginia	Electricity generation	140	43	- 4.9%	1.7	194 kg/yr
	Transportation	39	12			
	Commercial, Residential, Industrial (CRI)	150	45			
Indiana	Elec. generation	295	37	+13.6	5.2	154 kg/yr
	Transportation	154	19			
	CRI	352	44			
Illinois	Elec. generation	316	29	+12.3	11.1	98.3 kg/yr
	Transportation	311	28			
	CRI	465	43			
Ohio	Elec. generation	335	28	+11.6	10.7	110 kg/yr
	Transportation	257	22			
	CRI	589	50			
Total		3403			28.7	
Total due to Elec. generation		1086				
Average			32%	+ 8.15%		139.08 kg/yr

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[1] Source: F.R. Drysdale and C.E. Kalif. 1977. The Energetics of the United States: An Atlas. Brookhaven National Laboratory, N.Y. BNL-50501.

The introduction of NO_x control devices on automobiles may change the relative magnitude of transportation-produced NO_x to a lower percentage compared to coal-fired power plant emissions. This is more likely to be important by the year 2000, but may also be affecting present and 1985 proportions.

Although additional and more current data are needed on NO_x production from all sources, and on the influence of emission heights on dry deposition and reaction times, the proportion of NO_x entering summer O_3-forming reactions and attributable to electricity generation appears to range from a probable lower bound of 30% (slightly below the utility portion of all NO_x emissions) to an upper bound of 50% (assuming growing-season levels of NO_x scavenging, and some decline in auto emissions). With time, as automobile NO_x decreases even further, these proportions seem likely to increase, but at a rate that is unknown. For the purposes of this study, a range of 30-50%, with a more "probable" value of 40%, appears to be a more reasonable alternative to the view of "no effect".

THE POSSIBILITY OF FUTURE INCREASES IN O_3

One indication of recent trends in O_3 is available from the Council on Environmental Quality (1979) which analyzed violations of the O_3 standard throughout the United States. The number of non-California violations above 0.08 ppm has increased from 1973 to 1977 -- a 16% increase over the four-year period, or approximately 4% per annum (Figure VIII-6).

At the same time, recent trends in NO_x concentration measured by the U.S. Environmental Protection Agency (1977) have shown 62% of the monitoring sites had increases in NO_2 between 1970 and 1976 (Table VIII-3). If these changes occur in the presence of significant concentrations of hydrocarbons, increases in O_3 would be expected.

The power plant development scenarios in ORBES also project increases in NO_x emissions by 1985 and 2000. With all three 1985 energy use scenarios, NO_x emissions are expected to have increased slightly during the 1976 to 1985 decade. With Scenario 7, by 2000, utility NO_x would increase 66% over 1985 levels. The increase would be 16% under Scenarios 2 and 2d (Table VIII-4).

A projection of increases in O_3 from increases in precursors depends on the evidence of NO_x-limiting conditions in the region. Figure VIII-7 presents a possible historical trend and three futures for ambient O_3 concentrations during O_3 episodes in the Ohio River Basin. Oceanic concentrations (0.02 to 0.03 ppm) plus natural background formation due to natural levels of hydrocarbons and NO_x accounts for an additional 0.02 to 0.3 ppm which suggests a range of .03 to .06 fifty years ago. There is also evidence that plants are adapted to concentrations of O_3 as high as 0.05 ppm (Bennett et al. 1974), but vegetation shows increasing inability to function properly in the presence of O_3 above concentrations in the range of .06 to .08 (see previous sections).

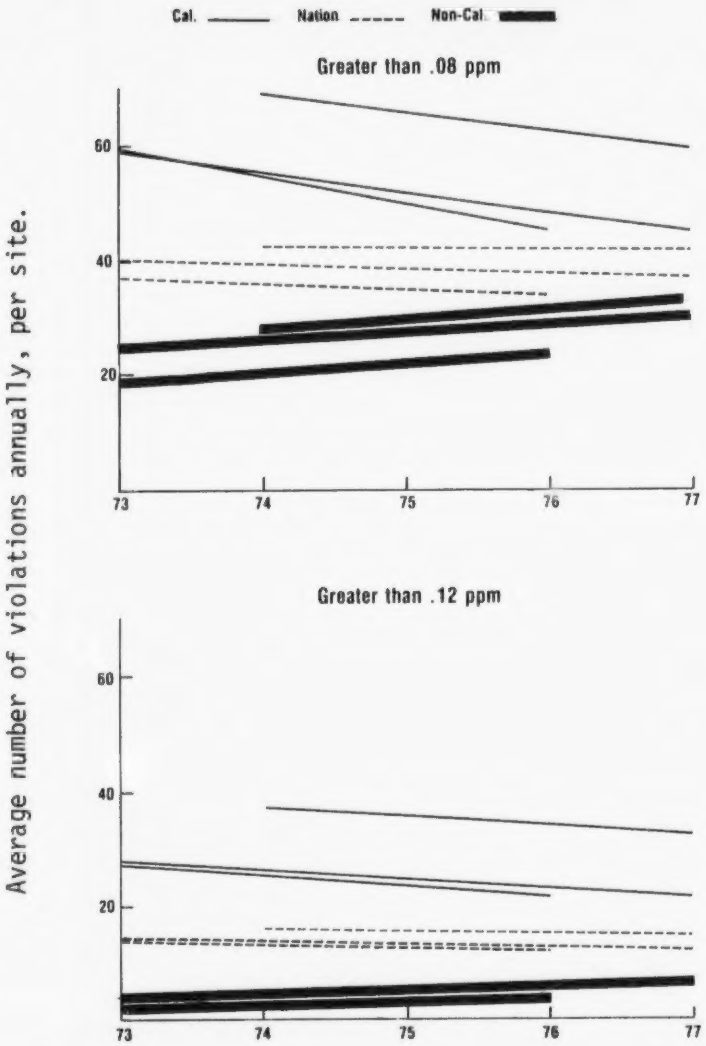

Figure VIII-6. Trends in O_3 standard violations at selected sites in California and the nation, 1973-1977. Source: Council on Environmental Quality 1979.

TABLE VIII-3. NITROGEN DIOXIDE TRENDS IN ANNUAL ARITHMETIC MEAN, 1970-76. Source: U.S. Environmental Protection Agency

Direction of Trend	California Sites	Non-California Sites	Total
Down	19	77	96
No change	4	12	16
Up	19	145	164
TOTAL	42	234	276

TABLE VIII-4. ESTIMATED NO_x EMISSIONS, O_3 CHANGES, AND PERCENT OF O_3 ATTRIBUTABLE TO POWER PLANT EMISSIONS IN THE ORBES REGION.

Year and Scenario	Total NO_x (10^9 kg/yr)	Utility NO_x (10^9 kg/yr)	% Utility NO_x	O_3 Change (ppm/10 yr)	% O_3 Due to Power Plant Emissions		
					Upper	Lower	Probable
1976	4	1.35	34%	0.020	50	30	40
1985 (All scenarios)	4.45	1.55	35%		50	30	40
2000 (7)[c]	4.71	2.58	55%	0.007	70	50	60
2000 (2[a] & 2d[b])	3.93	1.80	46%	-0.014 (from 1985)	60	40	50

[a] (2) - Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

[b] (2d) - Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

[c] (7) - Scenario assuming compliance with SIPs by 1985, high electrical growth, and extended power plant service life.

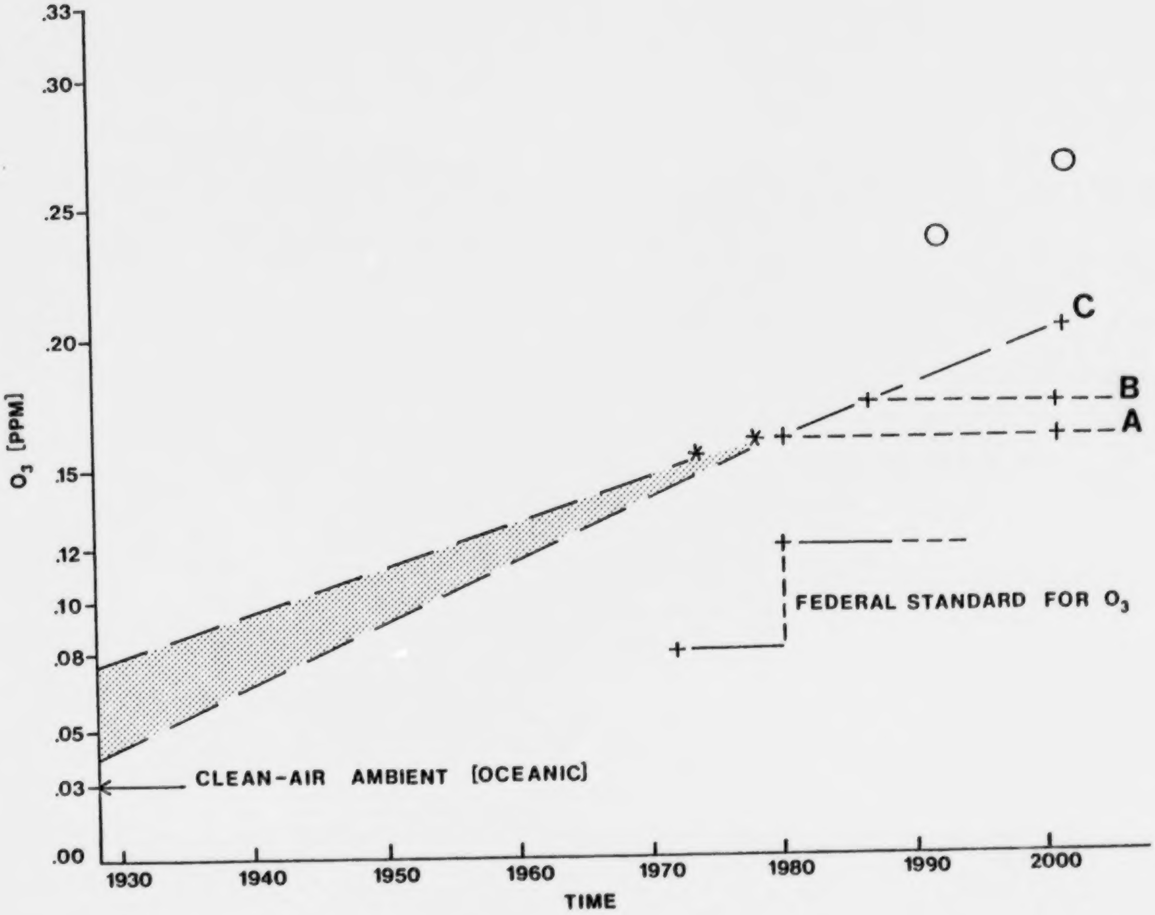

Figure VIII-7. Trend in episode ozone concentrations in Ohio River Basin from 1929 to 2000 A.D. (* = Highest hourly value for year at Indianapolis; o = Expected maximum hourly average at 1:1 $NO_x:O_3$ production rate.)

Although indirect evidence points to some occurrences of regional air pollution disease prior to the 1950s (Hayes and Skelly 1977, Johnson 1922, Burk and Heggestad 1956), oxidant-induced diseases on the traditional sensitive bioindicators (eastern white pine and tobacco) were essentially unnoticed, and the subject was of little interest in plant disease research until the mid-1950s when the symptoms began to be more frequent. By 1955 however, Burk and Heggestad (1956) estimated that a one million dollar loss to the cigar wrapper industry occurred in the New England states due to air pollutants, and by 1972, Weidensaul and Lacasse (1972) estimated an 11.5 million dollar loss to agriculture in Pennsylvania. A similar trend in O_3 effects has been documented in California, where Millican (1976) estimated a more than 300% increase in air pollution damage to agriculture in four years, from 16.1 million dollars in 1970 to 55.1 million dollars in 1974. The primary reasons given for such increasing estimates are (1) trans-decade increases of oxidant air pollution, and (2) air pollution-induced plant diseases on larger numbers of crops and forest species.

Thus, ambient O_3 appears to have increased gradually since before the 1950's until 1979 (Figure VIII-7). The average rate of change would be approximately 0.02 ppm per decade. Under limited growth scenarios, this trend in O_3 could level off. With NO_x abatement in the transportation sector, O_3 would be expected to level off by 1985 (trend B of Figure VIII-7), even with a doubling of energy-generating capacity from coal-fired power plants. However, with a larger increase in electricity generation by coal-fired power plants (without NO_x residuals control technologies) and approximate doubling of NO_x from the power plants, O_3 could continue to increase at its historic rate (trend C). In this latter scenario, increased NO_x from power plants is projected to overwhelm the potential decreases in NO_x from the transportation sector. Trend A, which assumes a leveling off of O_3 concentrations as of 1979 (due to implementation of state plans for pollution abatement), seems improbable because abatement initiatives are barely underway in most states and, in 1979, the Federal O_3 standard was reduced (see Figure VIII-7). Consequently, the historical trend in O_3 increase of about 0.02 ppm O_3 per decade appears to be a reasonable estimator of future trends, at least to 1985.

More serious problems develop in trying to project the status of total NO_x emissions by the year 2000 and their implications for O_3 concentrations. Scenario 7 diverges from the others between 1985 and 2000 (Table VIII-4), and the rate of increase in NO_x emissions is such that O_3 from 1985 to 2000 could increase considerably. Decreases in non-utility emissions also could be expected during this period and the full change in NO_x might be estimated as follows (all NO_x emissions are in units of 10^9 kg/yr):

(1) Planned decreases in mobile-source emissions from 1985 to 2000 may balance the increase in utility NO_x emissions of 0.25×10^9 kg/yr in Scenarios 2 and 2d (Table VIII-4). This leaves 0.78×10^9 kg/yr as the difference between increased utility emissions

in Scenario 7 and non-utility reductions projected (Table VIII-4).

(2) Of the projected 0.78×10^9 kg/yr increase in Scenario 7, two-thirds (0.52) could be abated by the further decreases in non-utility emissions due to less driving, smaller cars, and stationary source controls, leaving one-third (0.26) from utilities. This portion could be expected to cause a one-third rate of O_3 increase (0.007 ppm O_3 per decade) in comparison to the historic trend.

Accordingly, for Scenarios 2 and 2d, utility NO_x emissions are projected to increase by 0.25×10^9 kg/yr between 1985 and 2000 (Table VIII-4). Applying the same non-utility decreases as above ($0.25 + 0.52 \times 10^9$ kg/yr) suggests that total NO_x emissions would decrease by 0.52×10^9 kg/yr. This could produce a decrease in O_3 of 0.014 ppm per decade.

ESTIMATED UPPER AND LOWER BOUNDS OF CROP GAINS FROM OZONE ABATEMENT

The effects of O_3 on crops for the base year 1976 have been estimated from monitored O_3 exposures in the ORBES region and expressed as crop-loss functions (Table VIII-5). To estimate crop-loss functions for 1985, it must be either assumed that the 1976-77 O_3 concentrations will prevail (in which crop losses will be unchanged from the base year), or that an increasing (or decreasing) trend in O_3 concentrations will occur as discussed above. All three scenarios for 1985 have similar NO_x emissions, so all three would produce the same crop losses by either assumption. The evidence reviewed here suggests that an increasing trend of ozone is the more probable, within limits of some uncertainty.

Estimated crop-production losses due to O_3 in the Ohio River Basin also need to be understood as equivalent to prospective crop production gains resulting from emission abatement (Table VIII-6). As for SO_2, the probable clean air production for each crop was calculated as follows:

$$\text{Probable Clean Air Production} = \frac{\text{Harvested Mean Yield (1975-1977)}}{1 - \text{Probable } O_3 \text{ Crop Loss Function (1977)}}$$

All crop losses reflect the variable loss functions developed in Section V as a proportional loss from the "clean air" yield. The estimated total potential gains in production for all O_3 impacts under each scenario are listed in Table VIII-6. The 1976 base year losses for soybeans range from 27.7 to 144.2 million bushels, for corn from 84.7 to 317.8 million bushels, and for wheat from 5.5 to 18.3 million bushels. Changes in crop gains for the future depend on reductions or increases in NO_x as discussed above. Overall, potential gains from prospective abatement by 1985 are expected to increase to about twice the level of 1976, regardless of scenario. Crop losses for 2000 (equal to potential crop gains through abatement) vary, depending on which scenario actualizes. Crop losses will be considerably less under the compliance

TABLE VIII-5. CROP LOSS FUNCTIONS FOR OZONE EFFECTS ON THREE SPECIES IN THE OHIO RIVER BASIN (%).

Crop	1976	1985 (All 3 Scenarios)	2000 Scenario 7[c]	2000 Scenarios 2[a] & 2d[b]
Corn				
Upper bound	15	18	19	11
Lower bound	4	7	9	4
Probable	8	11	12	6
Soybeans				
Upper bound	26	41	47	20
Lower bound	5	24	30	2
Probable	14	28	34	7
Wheat				
Upper bound	10	18	23	8
Lower bound	3	5	7	2
Probable	6	13	14	3

[a] (2) - Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

[b] (2d) - Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

[c] (7) - Scenario assuming compliance with SIPs by 1985, high electrical growth, and extended power plant service life.

TABLE VIII-6. TOTAL GAIN IN PRODUCTION (10^6 BUSHELS) OF THREE CROPS IF OXIDANTS ORIGINATING FROM ALL SOURCES IN THE ORBES REGION WERE ABATED. (Minimum, maximum, and probable reflect differing oxidant loss functions--See text and Table VIII-3.)

Crop	Probable Clean Air Production 3-year average 1975-1977	Total Gains 1976	Total Gains 1985[a]	Total Gains 2000[a]	Total Gains 2000[b]
SOYBEANS	554.5				
Minimum		27.7	133.1	11.1	166.3
Maximum		144.2	277.3	110.9	260.6
Probable		77.6	155.3	38.8	188.5
CORN	2,118.6				
Minimum		84.7	148.3	84.7	190.1
Maximum		317.8	381.3	233.1	402.5
Probable		169.5	233.0	127.1	254.2
WHEAT	182.5				
Minimum		5.5	9.1	3.7	12.8
Maximum		18.3	32.9	14.6	42.0
Probable		10.9	23.7	5.5	25.6

[a] Gain occurring in 1985 using compliance (2), no compliance (2d), or high electricity growth (7) scenarios, and in 2000 using scenarios 2 and 2d.

[b] Gain occurring in 2000 using only high electricity growth (7) scenario.

(2) and no-compliance (2d) Scenarios, compared with the high electricity growth (7) projections in 2000. The high electricity growth scenario is estimated to produce losses to the three crops greater than estimated for 1985 (Table VIII-6).

The portions of these projected gains probably associated with oxidant abatement from coal-fired power plant NO_x residuals are estimated as 30% to 50% of total ORB oxidant-induced crop effects (Table VIII-7). The total base period oxidant effect is estimated to range between 35 and 40 million bushels with the greatest potential gain occurring on corn. Estimated benefits from power plant NO_x abatement by 1985 rise to a value between 87 and 320 million bushels for the three crops. Dramatic decreases in crop losses from power plants could occur by 2000 under scenarios 2 and 2d, but could continue to increase under scenario 7, with a quadrupling of generating capactiy (Table VIII-7). If the null hypothesis of no additional ozone above base year levels is real, annual losses to forests and crops equal to 1977 would be expected to continue until 2000.

Related information bearing on the estimation of crop losses includes knowledge of weather patterns during O_3 episodes and the mix of sensitive and tolerant crop varieties in the Ohio River Basin. Additional analysis using these data bases would be desirable. For the present, it is assumed that the corn, wheat, and soybean varieties in the Ohio River Basin are as sensitive to O_3 as those field varieties on which the crop loss functions were based. The projected losses to the three crops may tend to become production limiting factors, especially for soybeans, in the early 1980's even without visible injury occurring. Consequently, new O_3-tolerant varieties might be introduced, decreasing projected losses.

CONCLUSIONS

1. Ozone concentrations being monitored in rural regions of the Ohio River Basin are high enough, and are of sufficient duration, to cause negative responses in vegetation and crops.

2. Since pre-industrial ambient O_3 levels at low altitudes appear to have been low (0.03 to 0.04 ppm, except perhaps to 0.08 ppm during episodes of thunderstorm or cyclonic activity), natural events do not account for the high O_3 levels at low elevations during pollution episodes in the Ohio River Basin.

3. Under certain weather conditions, power plant emissions of nitrogen oxides, and reactions with non-methane hydrocarbons in the surrounding atmosphere from both natural and anthropogenic sources have been shown to produce as much as 20% to 30% increase in ambient O_3 concentration.

4. Intact plumes, or air parcels with accumulated stack emissions, can persist long periods and move long distances over land and water. In the Ohio River Basin, O_3, NO_x, and hydrocarbons are

TABLE VIII-7. CURRENT AND PROJECTED REDUCTIONS IN ANNUAL YIELD FROM OZONE/OXIDANTS ATTRIBUTABLE TO OHIO RIVER BASIN POWER PLANTS (10^6 Bu./Yr.)

Crop	Total Area (1,000's Acres)	1975-1977 Base Period	1985 (All Scenarios)	2000 (2[a] & 2d[b])	2000 (7)[c]
Soybeans	13,888				
Minimum		8.3	39.9	4.4	83.2
Maximum		72.1	113.7	66.5	182.4
Probable		31.1	62.1	19.4	113.1
Corn	18,715				
Minimum		25.4	44.5	33.9	85.3
Maximum		158.9	190.7	139.8	281.8
Probable		67.8	93.2	63.6	152.5
Wheat	4,211				
Minimum		1.6	2.7	1.5	6.4
Maximum		9.1	16.4	8.8	29.4
Probable		4.4	9.5	2.7	15.3
Totals	36,815				
Minimum		35.3	87.1	39.8	174.9
Maximum		240.1	320.8	215.1	493.6
Probable		103.2	164.8	85.7	280.9

[a] (2) - Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

[b] (2d) - Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

[c] (7) - Scenario assuming compliance with SIPs by 1985

present in significant concentrations over rural areas outside of plumes.

5. During the periodic episodes of stagnating air masses in the Ohio River Basin during the summer, O_3-forming reactions can take place over extended time periods utilizing NO_x and hydrocarbon concentrations in the order of .003 and .20 ppm, respectively.

6. Because of the presence of both naturally produced hydrocarbons from forests and agricultural crops in the Ohio River Valley, and the anthroprogenic sources in the region, the amount of hydrocarbons appear rarely to be a rate-limiting factor for production of O_3. NO_x, which is produced in the Ohio River Basin region from urban sources as well as power plants, appears to be a more frequent limiting factor for O_3 production.

7. Analysis of the rate limiting components suggests power plants in the Ohio River Basin probably contribute between 30% and 50% of the NO_x contributing to O_3-forming photochemical reactions.

8. Increases in O_3 seem likely to occur downwind from primary pollution sources in the Ohio River Basin because of the elevated levels of NO_x. The average episode-peak level of O_3 could continue to increase at about 0.02 ppm per decade if NO_x emissions expand as projected. If transportation emissions abate considerably, O_3 concentrations could decrease, despite some increases in utility NO_x emissions.

9. Crop loss functions in 1977, based on measured O_3 concentrations and calculated concentration-hours of exposure, range from a lower bound of 5% to an upper bound of 26% for soybeans, 4% to 15% for corn and 3% to 10% for wheat (Section V). Crop loss functions in 1985 and 2000 are based on projected O_3 concentrations, and, therefore, projected cumulative concentration-hours of exposure. Soybean yield reductions are estimated to range from a lower bound of 24% to an upper bound of 41% in 1985 (all scenarios); corn yield reductions would range from 7% to no more than 18%; wheat yield reductions should range from 5% to 18%. An assumption of high electrical growth (Scenario 7) leads to projected yield reductions in 2000 of from 30% to 47% for soybeans; from 9% to 19% for corn, and from 7% to 23% for wheat. Under Scenario 2 and 2d, the assumption of reduced non-utility NO_x emissions leads to a considerable reduction in projected yield losses due to O_3 by the year 2000: 2% to 20% for soybeans, 4% to 11% for corn, and 2% to 8% for wheat.

10. Estimates of total production losses for all three crops due to ozone are multiplied by the proportion of O_3 believed to be attributable to power plants to derive estimates of crop pro-

duction benefits associated with the power plant development options examined in ORBES. If the assumptions of power plant NO_x contributions to regional O_3 are accepted, the utility-related portion of estimated soybean losses range from 8.3 to 72.1 million bushels in 1976, increasing to a range of 39.9 to 113.7 million bushels in 1985, and from 83.2 to 182.4 million bushels under the assumption of high electrical growth by the year 2000. Corresponding estimates for corn (in millions of bushels) range from 25.4 to 158.9 (1976), 44.5 to 190.7 (1985), and 85.3 to 281.8 (2000); for wheat, the estimates are 1.6 to 9.1 (1976), 2.7 to 16.4 (1985), and 6.4 to 29.4 (2000).

11. These losses represent 1.5% to 33% of current production for soybeans, 1.2% to 13.3% for corn, and 0.9% to 16% for wheat. Within the limits of the data available, the projected losses for the three crops can be considered equivalent to potential crop production gains achievable from complete pollution abatement.

12. A relatively untested null hypothesis regarding the time-trend of O_3 concentrations in the ORBES region would show present O_3 exposures as unchanged from historic levels, and, therefore, unlikely to be influenced by current or future interventions in the emissions of O_3 precursors. While this possibility cannot be excluded, it appears improbable at this time. A *status quo* of present O_3 pollution would tend to assure crop losses as estimated for the base year.

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SECTION IX

ESTIMATING FOREST GROWTH LOSSES DUE TO OXIDANTS AND POLLUTANT-INSECT INTERACTIONS

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INTRODUCTION

Questions regarding the effects of ORBES region utility emissions on forests and other non-agricultural species require an approach much different from that applied to crops. The region has in the order of 50 native forest tree species, with many more introduced as horticultural plantings. These species are exposed year after year to atmospheric pollutants, and for the conifers, which usually maintain several years' foliage, effects are expressed cumulatively over years rather than in a single growing season. In addition, the "product" value is the accumulating woody growth or the plant survival (as in the case of horticultural material), rather than an annual harvest that can be removed from exposure each year.

Because of differences in duration of exposure, number of species, and limited experimental data base, it is difficult to differentiate effects on forests of primary gaseous emissions from those of secondary pollutants, such as oxidants and acid rain. The differences can be distinguished where one or another pollutant dominates, but over a large area, such as the Ohio River Basin, responses tend to become integrated. Also difficult to assess are the effects on trees from elevated insect populations which take advantage of weakened host plants over a period of years. Air pollutants also have effects on natural insect control organisms, such as birds and hymenopteran parasites which need to be considered. The following sections review the evidence on these influences and provide a preliminary estimate of the level of effects attributable to pollutants.

BACKGROUND ON FOREST GROWTH EFFECTS

The effects of air pollutants on forest growth have been documented best in the San Bernardino Mountains in southern California, a region of long-term exposure to ozone (O_3) at moderate to high concentrations. Here, ponderosa pine growth losses now approach 37% reduction in radial growth in trees under age 30 (see Figure IX-1), with annual mortality rates averaging nearly 3% in ponderosa and Jeffrey pine. Studies in the most affected forests indicate an 83% loss of merchantable volume growth of 30-year-old trees (Figure IX-2) (U.S. EPA 1977).

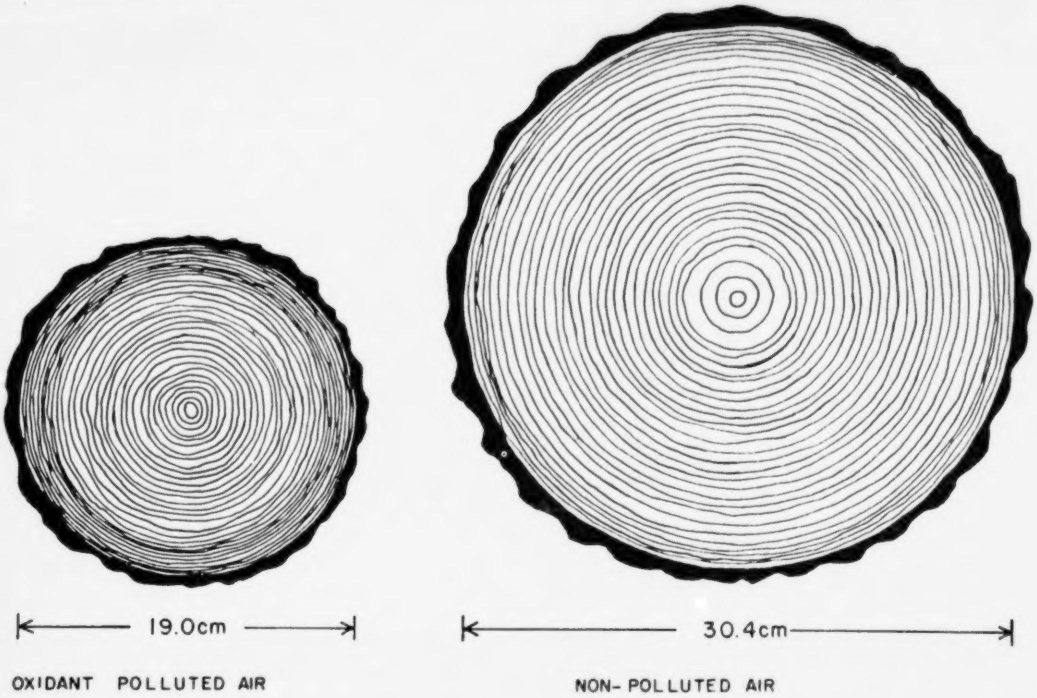

19.0cm
OXIDANT POLLUTED AIR
30.4cm
NON-POLLUTED AIR

Figure IX-1. Calculated average cross-sections of two 30-year-old ponderosa pines at breast height grown in polluted air (left) and in non-polluted air (right) based on radial growth samples from 1941-1971 and 1910-1940. (See U.S. EPA 1977)

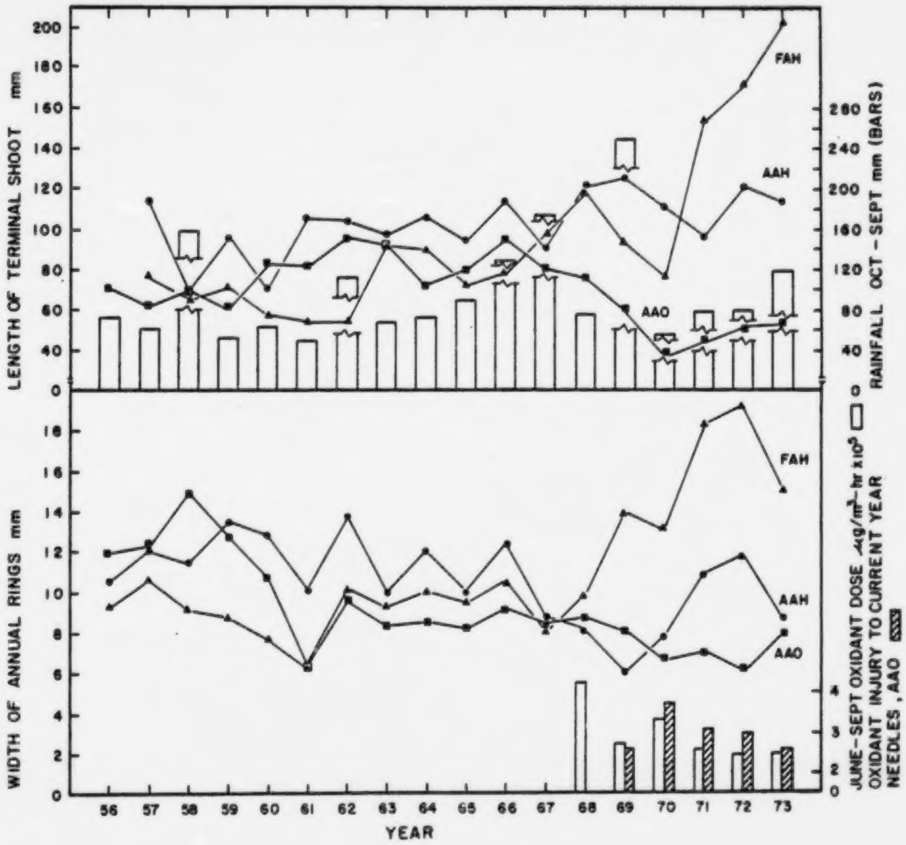


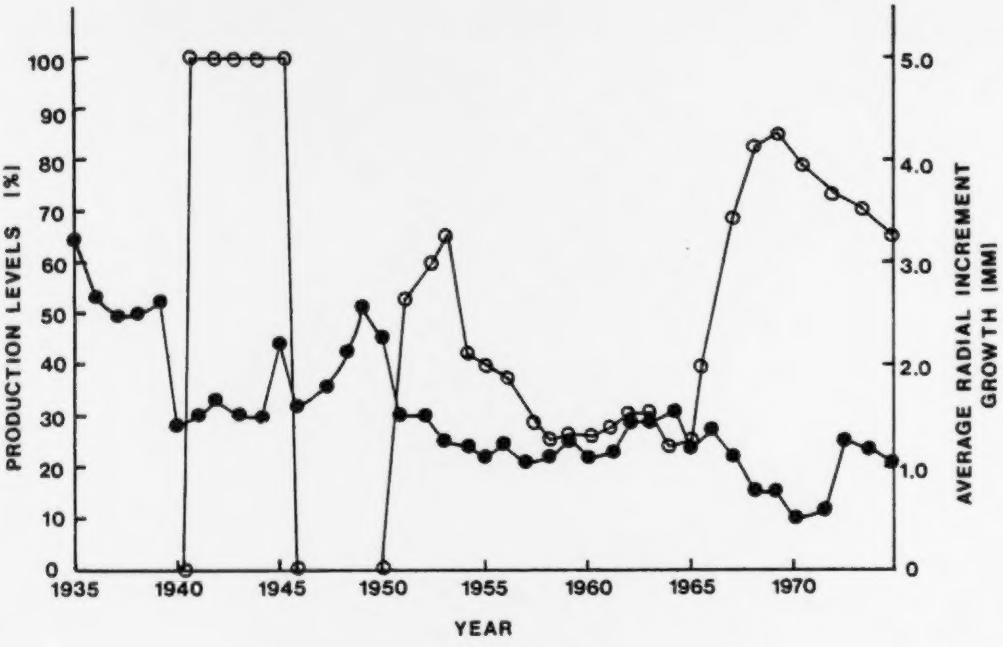
Figure IX-2. Relationship of precipitation, oxidant dose and foliar injury to radial and vertical growth of ponderosa pine saplings before and after treatments in filtered (FAH) or unfiltered (AAH) air greenhouse and an outside ambient air treatment (AAO). (See U.S. EPA 1977)

Forest growth losses in the eastern U.S. have been less well documented. Several studies have shown, however, that air pollutants markedly reduce annual growth ring widths. Skelly et al. (1979) studied the impacts of photochemical oxidants (including O_3) on eastern white pine in the Shenandoah National Park, Blue Ridge Parkway, and Great Smoky Mountains National Park. Diseased white pines were tagged at various locations and rated for disease severity over time. An 11% annual mortality rate was measured, a greater loss than has been observed in southern Californian yellow pine forests (U.S. EPA 1977). Since there is more photochemical smog in the southern California forests east of the Los Angeles air basin than in the eastern U.S., the eastern white pines may be more sensitive than western ponderosa pines.

In a more specific examination of direct gaseous pollution effects, Phillips et al. (1977a) studied eastern white pine stands near a munitions plant in southwest Virgina and noted significant growth losses with and without visible symptoms of air pollution disease. Severely affected trees averaged 66% of the height of asymptomatic trees. The annual radial incremental growth of yellow poplar also was reduced as annual ammunition production (and air pollution) levels went up during three wars (Stone and Skelly 1974). The observed decline in annual ring widths of the eastern white pines was not necessarily related to the occurrence of visible manifestations of disease (Figure IX-3), and growth losses were not associated with rainfall variations.

Phillips et al. (1977b) also studied growth fluctuations of loblolly pines due to wartime air pollution emissions. Again, a strong trend toward reduced annual ring widths was associated with the periodic air pollution emissions. A 45% reduction of growth was predicted by regression equations relating radial increment to production activity of an ammunition plant. The growth reduction was not necessarily associated with the appearance of visible symptoms of injury from the air pollutants.

Ring-width data from a series of trees downwind from a munitions plant in central Wisconsin are shown in Figure IX-4. Site 1 is closest to the plant (ca. 1 km), while Site 5 is farthest (ca. 30 km); Site 5 is 5 km from a coal-fired power generating station that began operations in 1975. The oldest trees at Site 2 show effects similar to those of Phillips et al. (1977a, Figure IX-3) for the 1943-45 period and the Korean War. Sites 1 and 2 (Figure IX-4) show the effects during the Vietnam War, but the remote Sites 4 and 5 also show small reductions in growth and partial recovery. The munitions plant emissions reduced the radial growth by approximately one-half during World War II, but afterwards the trees recovered fully. Since the closing of the munitions plant after the Vietnam War, the trees have not fully recovered, a pattern that is also evident in Figure IX-3 for the Virginia munitions plant. The data suggest a trend toward declining ring widths that is independent of tree age or known emission sources. The continuing increase in oxidant levels, as indicated in summertime O_3 peaks (see Section VIII), may explain this decline, in view of known sensitivity of pines to oxidants and the prob-



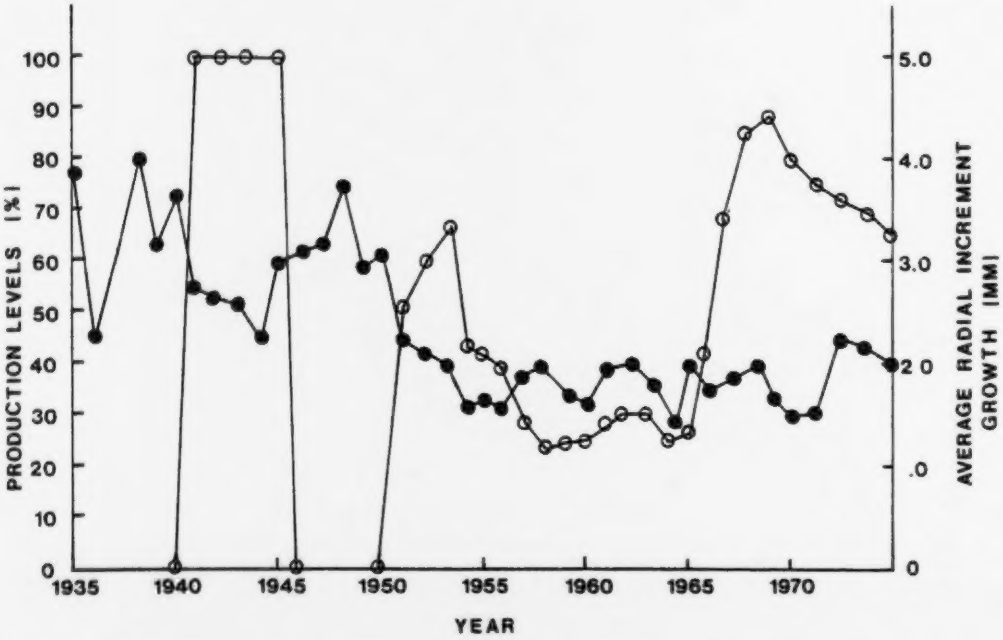


Figure IX-3. Relation of reduced ring widths of eastern white and loblolly pines to increased air pollution from a Virginia munitions plant over a 3-war period of 1935 to 1975. Source: Phillips et al. (1977a,b).

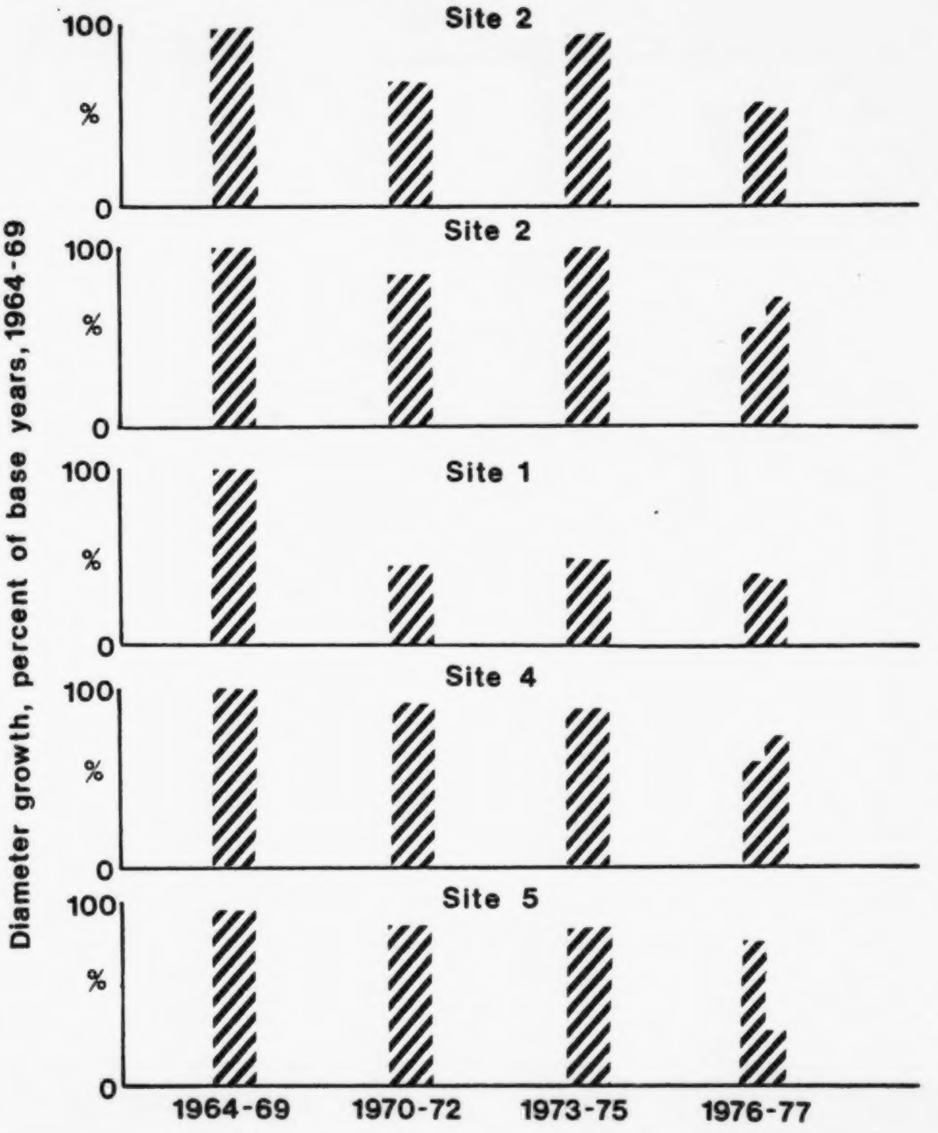


Figure IX-4. Ring-width of trees downwind from a munitions plant in central Wisconsin in four grouped time intervals from 1964 to 1977.

ability of synergistic responses to O_3 and low levels of SO_2 and NO_x. Additional research using larger numbers of increment borings, adjustments for climate and other effects, and local monitoring of pollutants are needed.

The results from the Indiana white pine foliage survey described in Section IV provide background on the levels of air pollution effects in the Ohio River Valley area. The spectrum of disease symptoms known to be usually associated with air pollutants was used to formulate "disease index" values for various locations in Indiana and Wisconsin. In the disease index, a healthy tree would receive a value of 80, with a maximum possible disease value of 400. Usher and Williams (Section IV) monitored white pines in 1980 for symptoms of air pollution disease at four relatively "clean" sites near the Columbia Generating Station at Portage, Wisconsin. These stands produced a mean injury index value of 162, indicating significantly less disease than at the Indiana sites. The differences in disease indices at the various locations correspond to observable canopy changes; when trees lose foliage prematurely, canopy density deteriorates, and the trees experience a decline in stem vigor.

ESTIMATING TOTAL FOREST GROWTH REDUCTION IN THE OHIO RIVER BASIN

The dominant forest species in the Ohio River Basin are oaks and hickories in the western part of the region, with larger amounts of beech, yellow poplar and sugar maple in the east. Conifers are sparse but have been planted extensively. The principal sensitive species in the region are eastern white pine, red pine, jack pine, Virginia pine, and several hardwoods, including black walnut, yellow poplar, sycamore, black cherry, and black locust. These species are native to most of the region and widely planted. The black oak group is not as sensitive as the species listed above, but is very common. Sycamore and yellow poplar are important high-production species over much of the region.

1976 Baseline Estimates and Rationale

One data base for estimating forest growth losses is that of Linzon (1971), who studied air pollution effects on eastern white pine for long distances downwind of the Sudbury, Ontario, nickel smelters. At Sudbury, impact zones have been described along a gradient of SO_2 damage downwind of the smelters. In the most remote, non-fume zone, the mean percentage of trees displaying symptoms of air pollution-induced foliar injury on current year foliage (1963) and 1-year foliage (1962) was 2.1% and 4.1%, respectively. At two study sites in the outer-fume zone, the mean percent of air pollution injury symptoms on current foliage was 1.1% compared to 11.9% on 1-year-old foliage. At three sites in the intermediate-fume zone the ratio was 0.6/25.3. To recapitulate, the mean ratios of percent of eastern white pines displaying foliar symptoms on current and 1-year-old needles were:

Non-fume zone	1.0/1.95
Outer-fume zone	1.0/10.82
Intermediate-fume zone	1.0/42.17

Measurements of annual wood volume increment on these trees showed that eastern white pine in the non-fume zone added from 0.02 to 0.09 cubic feet (ft^3) per year more than trees growing in the outer-fume zone. Mortalities from known and unknown causes were subtracted before estimating growth. Linzon reports that since net merchantable volume for sawlogs is estimated to be approximately 50% of gross total volume, a doubling of the loss is warranted (e.g., annual increments of 0.04 to 0.18 ft^3/tree/year). This loss occurs with a first- to second-year needle injury ratio of approximately 1:11.

Linzon did not report SO_2 or other air pollution concentrations in the outer-fume zone or the non-fume zone. However, SO_2 and ozone have been monitored in the vicinity of a coal-fired power plant near Portage, Wisconsin, where Karnosky (1980) measured the percentage of trees with tip necrosis on first-year needles. Usher (Section IV) visited four of Karnosky's plots at Portage and measured the percentage of air pollution-induced symptoms on 1-year-old needles. The ratio of percentage of tip burning and other symptoms induced by air pollution between current-year needles (Karnosky's data from 1971-1977) and 1-year-old needles (Usher's data in 1980) was 1:10.2. Assuming the growth potential and pollution sensitivity of white pine to be similar at Portage and at Sudbury, it appears that eastern white pines having similar leaf injury ratios due to air pollutants experience similar growth effects, regardless of geographic location.

In addition to foliage symptoms, chlorotic dwarfing is a severe manifestation of air pollution disease on eastern white pine. Dochinger and Heck (according to Mann _et_ _al_. 1980) demonstrated that chlorotic dwarf disease on eastern white pine can be produced by chronic exposure to SO_2 at levels as low as 0.01 ppm or 27.8 micrograms/cubic meter ($\mu g \cdot m^3$). At the Wisconsin study sites, chlorotic dwarf disease is observed occasionally, and other air pollution symptoms are common. Air pollution monitoring stations in Wisconsin (Table IV-6) show that the monthly average of SO_2 exceeded 20 $\mu g/m^3$ during only 5 months from 1973 to 1978, but O_3 was commonly present at concentrations above 0.10 ppm. This air pollution dose has been shown to be phytotoxic to white pine (Gerhold 1977). A combined pollutant load, which produces a first-year/second-year leaf injury ratio of 1:10, is therefore considered sufficient to cause an amount of wood loss per tree equal to that in the outer-fume zone at Sudbury (0.04 to 0.18 ft^3/year). Trees at all Indiana sites were significantly more severely injured than at the Wisconsin sites (Section IV).

Skelly (1980) reported a gradual incremental wood decrease in both tolerant and sensitive eastern white pines in the Blue Ridge Mountains of Virginia, where O_3 commonly exceeded 0.08 ppm. The accrued decrease in annual wood production rates from 1955 to 1978 was more than 40%, regardless of clone sensitivity. For sensitive genotypes, measurements of the decrease in annual wood increment showed the reduction had reached approximately 50% by the end of the 23-year period. Thus, the clean air yield could be at least 40% greater than is now realized for eastern white pine under these conditions. Skelly's (1980) trees (average age 51-56 years) appear to belong to Linzon's 7-to-12-inch diameter at breast height (dbh) size-class.

Estimation of Growth Losses for Softwoods

The normal forest growth rate for the ORBES region has been calculated from U.S.D.A. Forest Service (1978) published estimates to be about 40 ft^3/acre/year for all species. This annual yield applies to 37.3 x 10^6 acres of forest in the ORBES region.

Softwood species, including air pollution-sensitive white, Virginia and shortleaf pines, comprise 4.2% of the forests, and hardwoods make up the remainder. The sensitive softwood species are assumed to experience a reduction in annual volume increment of 0.04 to 0.18 ft^3/tree/year, as calculated above. The annual softwood growth also can be computed as 4.2% of the annual growth (40 ft^3), or 1.68 ft^3/acre/year. The U.S. Forest Service (1978) has estimated the stocking rate of live softwood trees of _9 inches dbh or greater_ in the Ohio River Basin as 1.093 softwood trees per acre on commercial timber land, including white pine, red pine, Virginia pine and short-leaf pine. These species are all considered air pollution-sensitive. An annual reduction of 0.04 to 0.18 ft^3 (on the 1.093 trees/acre) produces a loss estimate for softwoods of 0.044 to 0.197 ft^3/ac/yr, or 2.6 to 11.7% of the potential 1.68 ft^3/ac/yr. When loss rates for the softwoods are applied to the 37.3 million acres of forested land in the Ohio River Basin, the estimate of softwood growth reduction is from 2.76 to 7.33 million ft^3/year. This range encompasses Linzon's estimate of 5.6% per year and is near Skelly's estimate of 2% per year for sensitive eastern white pines and softwoods of equal sensitivity.

Estimation of Air Pollution Losses for Hardwoods in the Ohio River Basin

West _et al_. (1980) developed an information base for concluding that a majority of the eastern hardwood species can be divided into resistant and intermediate air pollution sensitivity classes. Fifteen species were considered resistant, ten species intermediate, and only three hardwood species were considered sensitive. For purposes of growth computations the three sensitive hardwood species--black walnut, black cherry, and black locust--were considered to experience decreases in annual wood increment equal to those of the softwoods, i.e., 2.6 to 11.7% per year. According to West _et al_. (1980), these three species constitute 10.7% of the deciduous forest hardwoods. Thus, the hardwood growth estimate (total growth minus softwood growth, or 38.32 ft^3/ac/yr) can serve to estimate prospective growth reductions on hardwoods:

a) Total hardwood growth (38.32 ft^3/ac/yr) x sensitive hardwood growth (10.7%) gives a 4.1 ft^3/ac/yr growth of sensitive hardwood.

b) Each 4.1 ft^3 experiences a decrease of from 2.2 to 11.7% per year, or 0.09 to 0.48 ft^3/ac/yr loss.

c) When these growth rate reductions are multiplied by the total ORB forested acreage (37.3 million acres), the air pollution-induced growth loss to sensitive hardwoods can be projected at 3.36 to 17.9 million ft^3 per year.

Of the remaining hardwoods, ten species are classified as having intermediate sensitivity to air pollution. West *et al*. (1980) considered these species to be experiencing one-half as much wood loss as the sensitive species. This would mean an annual loss of from 1.1% to 5.9% (cf. b above), or from 0.02 to 0.09 ft^3/tree/yr. These ten hardwoods constitute 34.5% of the hardwood species under consideration (West *et al*. 1980). Thus, the cubic foot decrease in growth from hardwoods of intermediate sensitivity can be estimated from total wood accretion per acre per year (minus softwood growth and sensitive hardwood growth):

a) 34.5% of the total growth (40 ft^3) minus (1.68 ft^3 + 4.1 ft^3), gives 11.8 ft^3 of growth on intermediately sensitive hardwoods.

b) Assuming one-half as much loss to air pollution as in the sensitive species (a range of 1.1 to 5.9%), gives 11.8 x 0.011 to 0.13 ft^3/ac/yr, and 11.8 x .059 to .069 ft^3/ac/yr of growth reduction.

c) Thus, the total estimated loss to hardwoods of intermediate sensitivity in the ORBES region, on 37.3 million acres, would range from 4.85 million to 25.7 million ft^3 per year.

The remaining hardwoods are classified as resistant to air pollution and are not considered to be experiencing a reduction in growth rates.

Summary of Softwood and Hardwood Growth Reductions for 1976

The estimated total forest growth reduction from air pollution in the ORBES region, including sensitive softwoods, sensitive hardwoods, and intermediate hardwoods can be summarized as follows:

	Range, 1976 (10^6 ft^3/yr)
sensitive softwoods	2.76 - 7.33
sensitive hardwoods	3.36 - 17.9
intermediate hardwoods	4.85 - 25.7
resistant hardwoods	0.0 - 0.0
Total estimated reduction	10.97 - 50.93

The total annual wood production in the Ohio River Basin forests has been estimated by the U.S.D.A. Forest Service (1974) at 1.49 x 10^9 ft^3 of wood (40 ft^3/acre/yr x 37.3 million acres). The 1976 loss from air pollution stress on all species, therefore, would be estimated to have been from 0.7 to 3.4%, attributable to a combination of O_3 and SO_2 effects.

ESTIMATED FOREST LOSSES FOR THE YEAR 1985

To estimate potential reductions in forest growth for 1985, a relationship to the projected increased concentrations of O_3 (see previous sections) must be analyzed. As one approach, the 1976 loss function for forest species could be increased by an average 85%, in the same proportion as the estimated average increase in oxidant losses to all crops (corn, 37.5%; soybeans, 100%; and wheat, 117%). Because of the large number of forest species that are sensitive to ambient ozone levels, it appears possible that increases in wood losses will be at least as great as the mean increasing loss coefficient projected for these three agricultural crops. The relatively large increase between 1977 and 1985 (Fig VIII-17) occurs because of the logarithmic phase of the dose-response curves. A similar response also was postulated for oxidant-sensitive forest species in the southern Sierra Nevada Mountains of California (Freeman 1976). Since the projected increase is a mean estimate, it would include increased losses to trees of intermediate sensitivity but not to tolerant species.

The forest losses from oxidants for 1985, therefore, are estimated to be 1.3% to 6.3% of the total wood production for forest species (Table IX-1) for all 3 scenarios.

ESTIMATED FOREST LOSSES FOR THE YEAR 2000

In the year 2000, NO_x emissions are projected to decrease or increase, depending on which energy use scenario actualizes (see previous Sections). With the "business as usual" (Scenario 2) and "non-compliance" (2d) scenarios, the mean crop losses for corn, wheat, and soybeans due to ozone effects will decrease to 5.33% by the year 2000, compared with mean 1976 losses of 9.33%--a 57% decrease. Crop losses due to SO_2 by the year 2000 will vary under the scenarios, but will average 1.8% of the combined pollutant-induced losses. The total estimated losses in forest productivity by the year 2000 for scenarios that project doubled electricity generation (2 and 2d), therefore, range from 0.4 to 1.9% (Table IX-1); 57% less than in 1976.

With the high electricity growth scenario (7), the estimated O_3 losses will increase considerably for corn, wheat, and soybeans because of the projected increase in O_3, but SO_2 losses relative to the 1976 base year will decrease. Estimated forest losses due to oxidants under the high energy growth scenario for the year 2000 will be 171.9% greater than in 1976. This increase represents 29.83 to 138.5 million ft^3 of wood production lost, or 2.0% to 9.3% of total production. The range in wood losses attributable to power plants (30% to 50%) are listed in Table IX-1.

TABLE IX-1. TOTAL FOREST GROWTH REDUCTION IN THE OHIO RIVER BASIN DUE TO AIR POLLUTANTS, EXPRESSED IN MILLIONS OF CUBIC FEET OF WOOD. TOTAL NORMAL YIELD FOR THE REGION IN 1976 WAS 1.49 x 10^9 CUBIC FEET.

	Minimum Estimate	Maximum Estimate
1976		
Loss Functions (%)	0.7	3.4
Total Losses (10^6 ft^3)	10.97	50.93
Total Losses Due to Power Plants	3.29[a]	25.5[b]
1985		
Loss Functions (%)	1.3	6.3
Total Losses (all scenarios)	19.4	93.9
Total Losses Due to Power Plants	5.8	46.9
2000		
Loss Functions (%) - Scenarios 2 and 2d	0.4	1.9
Total 2 and 2d Losses	6.25	29.0
Total 2 and 2d Losses Due to Power Plants	1.87	14.5
Loss Functions (%) - Scenario 7	2.0	9.3
Total Scenario 7 Losses	29.83	138.5
Total Scenario 7 Losses Due to Power Plants	8.95	69.3

[a] = 30% of minimum total loss

[b] = 50% of maximum total loss

These loss estimates are dependent on the assumption that all of the hardwood growth rates are about equal, independent of pollutant sensitivity class. The dominant hardwood species may change during forest succession, but changes in forest succession rates may occur in the presence of pollutants that effect some species more than others (West et al. 1980, Kozlowsky 1980), and growth losses could be greater than estimated. It is important to consider that forest tree species reduced in populations or biomass by air pollution stress are not necessarily replaced by other species of similar physiognomy. Selection against pollution-sensitive tree strains will not necessarily favor filling those niches by air pollution-tolerant tree species (Farrar et al. 1977, Guderian 1977). Moreover, exposure of deciduous species to SO_2, even at quite low concentrations, may reduce wood quality (Keller 1980) or the silvicultural aspect of tree species (Keller 1978), thereby possibly rendering impacts greater than described here.

ESTIMATING THE EFFECTS OF INSECT/AIR POLLUTION INTERACTIONS

Insect damage has long been observed to be more prevalent in pollutant-impacted ecosystems. Insect pests attack air pollution stressed forests more than pollution-free forests and woodlands, and even normally innocuous insect pests may cause plant mortality. Although air pollution alone can kill forests, insects, root pathogens, and oxidants form an integrated complex in southern California that speeds up forest decline (U. S. Environmental Protection Agency 1977, James et al. 1980). Bark beetle populations in the southern California forests are reduced by pollution, but enough beetles remain to kill trees. The attack rate of bark beetles in oxidant stressed forests may be exaggerated (U. S. Environmental Protection Agency 1977) because the beetles are opportunists and attack trees with incipient diseases and dysfunctions caused by other factors. However, severely pollution-damaged trees have more hatched beetle eggs per length of gallery than less stressed trees.

Pollution downwind from point sources has been associated by Carlson (1975) and Carlson et al. (1974) with increased insect outbreaks causing tree mortality. Generally, insect damage increased as pollution concentrations increased. Other smog-induced or supported insect epidemics downwind from point sources of pollution have been observed in Europe and North America (Bromenshenk 1978, Heagle 1973). Bromenshenk (1978), Heagle (1973), and Glass and Loucks (1979) consider increased insect infestations in air pollution stressed ecosystems a natural continuum of decline induced by pollution that often ends in destruction of crops and forests by insects. Trees injured or weakened by pollutants are more likely to be attacked by insects that normally require weakened trees for successful reproduction (Heagle 1973). Also, foliage feeders appear to be more sensitive to air pollution than other insects (Heagle 1973). Some insect species may be stimulated by air pollutants (Heagle 1973).

A major factor in increased insect infestations may be that natural predators and parasites avoid air pollution (Bromenshenk 1978, Heagle 1973, Hillman 1972, Przybyski 1979). Without these predators and parasites, which function as natural controls for destructive insect species, outbreaks occur. Bromenshenk (1978), in a field fumigation study at Colstrip, Montana, observed that several species of predaceous ground beetles migrated out of polluted areas. This avoidance response was apparent at an SO_2 dose as low as 2 parts per hundred million (pphm) over the growing season. In work reviewed by Heagle (1973), insect parasites on the larvae of European pine shoot moths were fewer in areas where trees were severely affected by SO_2, even though larvae were abundant. Przybyski (1979) noted that aphids and true bugs flourish in environments polluted by combustion gases. He noted decreases in predatory pollution-sensitive Coccinelid beetles, which permitted increases in aphid populations near roadways. Increased aphid and scale populations in pollution-stressed forests have been reported elsewhere (Carlson 1975, Glass and Loucks 1979, Williams 1976).

Decomposer insects which function in the remineralization of forest litter and nutrients also can be affected by air pollution. Ginevan and Lane (1978) reported that the fruit fly was detrimentally affected by low levels of SO_2; the development time of larvae increased, and population productivity was reduced. Bromenshenk (1978) observed that decomposer beetles in field plots had lower populations compared with controls when exposed to chronic SO_2 pollution as low as 2 pphm. Carrion beetles were also fewer with SO_2 doses greater than 2 pphm.

Microarthropods other than insects also have important decomposer roles in nutrient cycling and litter accumulation. They, too, are sensitive to air pollution. In the San Bernardino National Forest, where O_3 is the principal pollutant, total numbers of microarthropods decreased by 38% in moderately pollution-damaged ponderosa pine stands when compared to very slightly damaged areas (U. S. EPA 1977). The total number of litter-dwelling insects did not vary appreciably, whether trees were "moderately" or "severely" damaged by air pollution. Thus, certain arthropod segments of the ecosystem fauna may be more pollutant-sensitive than insects. Mortality factors that cause on the order of 5-10% death rate increases during life cycles are known to have significant effects on insect population dynamics.

Honeybees are a particularly sensitive bioindicator of the effects of air pollutants. Bromenshenk (1978) found that bees are especially good indicators for the uptake of atmospheric flouride and sulfur. Honeybees have been observed to be directly affected (i.e., as avoidance) by coal-fired power plant emissions. In Pennsylvania and Europe (Bromenshenk 1978, Hillman 1972), for example, bees were killed by SO_2 and arsenic emissions. Bees and other pollinators are foragers and have territories that extend over large areas, suggesting that they would experience cumulative, rather than one-time exposures. Fluorides are toxic to honeybees even at low concentrations. Any interference with pollination mechanisms caused by air pollutants would be felt throughout ecosystems, especially in yields of forage crops, fruits, nuts, and vegetables, and in the rate of forest stand reestablishment.

Some insects have a high natural resistance to air pollutants, and are at a competitive advantage to displace other species in areas newly subjected to pollution. Ginevan and Lane (1978) state that the magnitude of the response observed with fruit flies strongly suggests that SO_2 may have ecologically significant effects on natural insect populations. Industrial melanism in moths from polluted woodlands and its reversal with pollution abatement, a classic example, is proof of the ability of some insects to change rapidly in response to air pollution. There is no evidence as yet that the insect control (parasite-predator) species have evolved such tolerance.

In areas chronically exposed to air pollutants at doses that are low, yet sufficient to elicit growth losses (e.g., in the Ohio River Basin), it is doubtful that the pollutants themselves will be the direct cause of tree mortality. However, Skelly's observation of a high incidence of *Verticicladiella* root rot on oxidant-stressed eastern white pines, James *et al.*'s (1980) determination of a direct association of *Fomes annosus* root rot with smog-damaged ponderosa pines, and Stark *et al.*'s (1968) observations of a close relationship between smog-damaged ponderosa pine and mortality-inducing bark beetle infestations strongly suggest that forest mortalities are caused by an integrated syndrome of pollutants, pests and pathogens, not merely from one causal agent.

The only estimator of annual losses where pollutant/insect/pathogen effects are indirectly implicated comes from long-term measurements of tree mortality. U.S. Forest Service estimates show that mortality in north-central and northeastern forests increased from 1.0 billion ft^3 in 1962 to 1.3 billion ft^3 in 1970 (U.S. Forest Service 1974), while mortality in southern forests increased from 1.1 to 1.2 billion ft^3. Inventory records show that forest mortality did not change in western and Pacific forests during the same period, despite the fact that these forests tend to be more over-mature. Thus, this recently induced mortality appears not to be a function of increasing forest age, although differences in inventory methods can produce data anomalies.

Many explanations are possible; one argument is that much of the regional decline is due to gypsy moth and bud worm infestations; outbreaks of blister, limb and gall rusts; vascular wilt diseases on oaks, maples, and elms; and a tendency for monocultures of tree species, especially in the South. With the present data base, it is not possible to quantify interactions of air pollution and these other agents of forest growth loss in the Ohio River Basin. However, there can be little doubt that such interactions exist and contribute to accelerated decline and mortality of trees. The absence of any estimate of insect/pathogen losses can be viewed as creating a major conservative element in the previous estimates of growth reduction.

LOSSES OF HORTICULTURAL PLANTINGS

To estimate direct losses of sensitive horticultural materials in urban areas, preliminary observations have been made in the Indianapolis area. Residential losses of major plantings, including mature conifers, shrubs, and garden plants on which air pollution injury symptoms are observable annually. These declines are not all directly attributable to air pollution injury, but all of this vegetation has been weakened by the air pollutants. Not all properties are affected every year, but nearly every residence loses some material once every decade. Although the final cause of mortality is usually insect outbreaks or disease, as with forests, the weakening of urban horticultural and landscape plantings by air pollutants appears to be a crucial factor in the etiology of plant loss. To estimate precisely the horticultural loss, a total urban population study would be required. Rural areas and small towns are not considered in the following estimate.

The number of urban residential properties in the Ohio River Basin is estimated to be about one million. Two-thirds of these properties are judged to have significant plantings. Thus, a conservative estimate is that about 0.7 million properties in the ORBES area have horticultural plantings of some significance.

In a recent insurance settlement in Indianapolis, one large specimen tree had an assessed value of $6,000. Other estimates suggest that mature trees on residential properties have an average value of $4,000. We have estimated the loss of such trees due indirectly to air pollution as one tree per 200 properties each year. Fully established small trees and shrubs are valued at $1,000 each, with one property in 50 losing a plant of this type each year. Garden plants and flower losses are estimated at $20 per year per property. The calculation of total horticultural losses is summarized in Table IX-2. Total losses are assumed to remain unchanged in future years, since pollution-sensitive varieties may be replaced by more resistant ones over the next several decades.

CONCLUSIONS

1. Ozone and SO_2 concentrations monitored in rural regions of the Ohio River Basin are sufficiently high and of sufficient duration to cause negative responses to forest growth in the region.

2. Forest growth and disease studies in Ontario, Central Wisconsin, Virginia, and Indiana indicate that reductions in forest growth in comparison to "clean air" growth rates are in the order of from 0 to 10% or higher depending on individual forest tree species sensitivity to air pollution.

3. Air pollution-sensitive conifer and hardwood species, including those of highest timber value, are estimated to be experiencing reductions of 2.6 to 11.7% in annual growth increment, excluding annual forest mortality. Total reduction in growth rates for the 1976 base year is estimated at 10.97 to 50.93 million ft^3 of wood.

Table IX-2. TOTAL HORTICULTURAL LOSSES IN THE ORBES REGION DUE TO POLLUTANT AND INSECT/DISEASE INTERACTIONS

	Population/Count	Value
Total Urban Population	4.1×10^6	
Total Residential Properties	1.0×10^6	
Total Properties with Plantings	0.7×10^6	
Ann. Loss of large trees per 200 properties	1.0	$4,000
Ann. Loss of small trees per 200 properties	4.0	$1,000 ea.
Shrubs & garden plants ($20 per property).		
Annual loss per property		$ 60
TOTAL ANNUAL LOSS		$42 × 10^6
Lower Bound, @ 30%*		12.6×10^6
Probable, @ 40%		16.8×10^6
Upper Bound, @ 50%		21.0×10^6

* Proportion of total losses probably attributable to coal-fired electric generating capacity in the ORBES region

4. Growth rates of hardwood species of intermediate air pollution sensitivity are estimated to have declined from 4.85 to 25.7 million ft^3 of wood in the Ohio River Basin forests in 1976.

5. Pollution-resistant species are not considered to have experienced growth losses to air pollutants in 1976.

6. Due to projected increases in O_3 concentrations, growth reductions are expected to double between 1976 and 1985, from 1.3 to 6.3% for all forest growth under the three development scenarios discussed. Reductions in wood production for 1985 are estimated to range between 19.4 and 93.9 million ft^3.

7. Losses to forest growth in 2000 for the 2 and 2d scenarios are expected to be less than in 1985, with loss functions ranging from 0.4 to 1.9%. Total forest growth losses for the high energy growth scenario (7) are estimated to range from 2.0 to 9.3% annually.

8. The estimated contribution of power plant emissions to the ozone levels (see Section VIII) suggest that between 30% of the minimum loss estimates and 50% of the maximum loss estimates, are associated with utility operations.

9. The amounts of wood production lost annually to air pollution/insect/pathogen interactions cannot be fully quantified with the current data base for ORBES forests, but the repeated observation that pathogens and pests cause ultimate mortalities on smog-stressed trees verifies their role and supports a view that the growth reduction estimates presented here are conservative.

10. Total annual losses to horticultural and landscape plantings in the Ohio River Basin are estimated to range from $12.6 million to $21 million.

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SECTION X

EPILOGUE: ESTIMATING THE AGGREGATE EFFECT OF AIR POLLUTANTS AND ACID DEPOSITION

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The goal of the studies reported in the previous sections has been to evaluate biological responses to present and proposed electric utility coal utilization scenarios in the Ohio Valley region. Effects on productivity of the agricultural and forest resource systems have been the principal measure. Thus, the results up to this point have been shown in various chapters as loss coefficients, bushels (for grain crops), cubic feet (for forests), and numbers of shrubs or trees (for horticultural materials).

One additional task was to consider the effects of acid rain believed to be associated with the emission changes projected for each utility development scenario. The purpose of this Epilogue is briefly to consider, in a qualitative way, the aggregate of both biological responses to air pollutants and the long-term physical and ecological responses to atmospheric deposition originating with precursors released in the Ohio Valley.

AGGREGATE EFFECTS OF THE AIR POLLUTANTS

Probably the only form in which the various effects described in the previous sections can be summarized is in monetarization. A variety of economic evaluations are included in reports being prepared for ORBES by Dr. W. Page of West Virginia University. Readers should obtain these reports for a full discussion of the economic significance of the crop losses or yield benefits.[1] The crop production gains expected from SO_2 abatement by power plants are summarized in Table X-1. In all cases these projected gains represent less than 1% of current production of each crop. The production gains for 1976 are partially achieved in all development scenarios, even when no compliance with pollution control laws is assumed. This is largely due to the replacement of old plants having high emissions with new, cleaner plants.

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[1] Page, W.A. 1980. "An Economic Analysis of Losses in Agricultural Production from Air Pollution in the Ohio River Basin Energy Study Region". Office of Research and Development. U.S. Environmental Protection Agency. In Preparation.

TABLE X-1. ESTIMATED CROP YIELD BENEFITS EXPECTED FROM ABATEMENT OF SULFUR DIOXIDE EMISSIONS FROM COAL-FIRED ELECTRICAL GENERATING STATIONS, 1976, AND NO-COMPLIANCE SCENARIO, 1985 AND 2000. GAIN IS FOR ENTIRE ORBES REGION.

		No Compliance Scenario[a] (2d)	
Crop	Estimated Crop Gain(10^3Bu) 1976	Estimated Crop Gain(10^3Bu) 1985	Estimated Crop Gain(10^3Bu) 2000
Soybeans			
Minimum	1,472	592	404
Maximum	2,124	1,923	1,304
Probable	1,472	1,331	901
Corn			
Minimum	None	None	None
Maximum	3,529	3,240	2,416
Probable	1,395	1,286	1,008
Wheat			
Minimum	214	225	160
Maximum	479	503	358
Probable	374	393	280

[a] Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

TABLE X-1. (continued) ESTIMATED CROP YIELD BENEFITS EXPECTED FROM ABATEMENT OF SULFUR DIOXIDE EMISSIONS FROM COAL-FIRED ELECTRICAL GENERATING STATIONS, COMPLIANCE SCENARIO AND HIGH-ENERGY GROWTH SCENARIO, 1985 AND 2000.

	Compliance Scenario[a]		High Energy Growth Scenario[b]	
Crop	Estimated Crop Gain (10^3Bu)1985	Estimated Crop Gain (10^3Bu 2000	Estimated Crop Gain (10^3Bu)1985	Estimated Crop Gain (10^3Bu)2000
Soybeans				
Minimum	255	187	252	220
Maximum	831	604	813	727
Probable	578	417	568	507
Corn				
Minimum	None	None	None	None
Maximum	1,341	1,007	1,321	1,274
Probable	521	400	520	453
Wheat				
Minimum	74	61	75	69
Maximum	166	135	167	156
Probable	129	107	131	121

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[a] Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

[b] Scenario assuming compliance with SIPs by 1985, high electrical growth, and extended power plant service life.

As noted in Section VIII, inter-regional transport of O_3 must be considered when interpreting the results in Table X-2. The indicated losses occur on the crops grown within the ORBES boundaries, but some part of the effects are induced by O_3-forming precursors emitted upwind of the region. At the same time, O_3-forming precursors originating in the ORBES region produce effects on crops outside the ORBES region. Since the O_3 concentrations along the ORBES region boundaries are not substantially different upwind and downwind of the region, and the elapsed time for air movement across the region is sufficient for most of the atmospheric reactions to be completed, crop losses downwind of the region--but attributable to ORBES region emissions--are assumed to be equivalent to crop losses inside the region, attributable to emissions upwind of the region. The proportion of the O_3-induced losses to be attributed to the electricity generating sector (40 to 50% for Scenarios 2 and 2d, and 50 to 70% for Scenario 7) has been examined at length in Section VIII.

The combined effects of O_3 on crops as well as forest growth and horticultural materials can be brought together only qualitatively, at least for the present. The value of the forest product loss is difficult to express. A reduction in forest growth, or an increase in mortality, represents an unrealized potential yield in an unharvested form. The usual market process is through sale of standing stumpage, a valuation that varies widely, depending on species and accessibility. Despite the relatively large volumes of apparent forest losses, the value attached to unharvested wood is low enough that total forest product values remain small.

The data in Tables X-1 and X-2 show that the direct effects of SO_2 emissions are modest relative to the effects from O_3, mostly due to the larger area affected by ozone. Indeed, the trade-offs between sulfur oxides (SO_x) and nitrogen oxides (NO_x) emission control costs and projected benefits appear to need re-evaluation.

Another dimension of the O_3 concentrations being observed, and projected for 2000, is its implication for carbon exchange. Reduced crop production and wood accretion because of ozone suppression would reduce carbon dioxide (CO_2) uptake. The recent trend in O_3 concentrations, estimated to have been from between 0.03 and 0.08 ppm during episode peaks in rural areas of the central United States in the 1920's, to recent values of 0.15 to 0.20 ppm, are summarized in Section VIII. The consequences of increasing ozone for vegetation and for CO_2 uptake and release are shown in Figure X-I for soybean responses. Soybean losses of 14%, in comparison to "clean air yields" were estimated in southern Indiana in 1977. Projections of increased ozone concentrations for 1985 and 2000 indicate probable soybean losses between 24 to 41% and 2 to 47%, respectively. Such responses by sensitive crop species are already known for California, where the development of somewhat resistant strains is in progress.

TABLE X-2. ESTIMATED RANGE OF TOTAL LOSSES FOR THREE CROP SPECIES DUE TO OZONE AND ASSOCIATED OXIDANTS FORMED FROM ORBES POWER PLANT EMISSIONS, 1977, 1985 AND 2000

Crop	1977 Yield Reduction (10^6 Bu.)	All Scenarios, 1985[a] 2 and 2d, 2000[a] Yield Reduction (10^6 Bu.)	7, 2000[a] Yield Reduction (10^6 Bu.)
Soybeans			
Minimum	8.3	39.9	66.5
Maximum	72.1	113.7	156.4
Probable	31.1	62.1	94.3
Corn			
Minimum	25.4	44.5	76.3
Maximum	158.9	190.7	241.5
Probable	67.8	93.2	127.1
Wheat			
Minimum	1.6	2.7	5.1
Maximum	9.1	16.4	25.2
Probable	4.4	9.5	12.8

[a] (2) - Scenario assuming compliance with SIPs by 1985 and doubling of generating capacity by 2000

(2d) - Scenario assuming no compliance with SIPs and doubling of generating capacity by 2000

(7) - Scenario assuming compliance with SIPs by 1985, high electrical growth, and extended power plant service life.

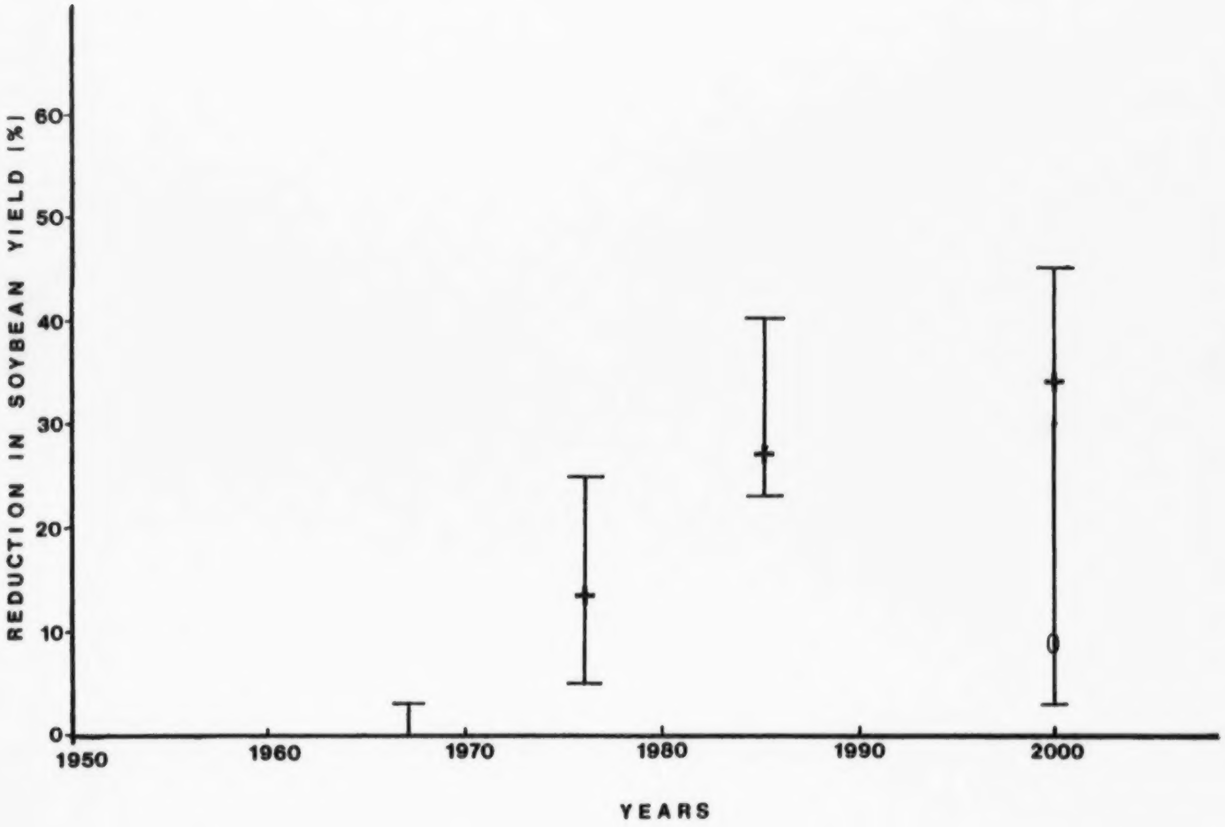


Figure X-1. Reduction in soybean yield (%) projected for the Ohio River Basin under conditions of high electrical growth. Horizontal bars represent maximum, minimum, and probable percent yield losses. Circle at 2000 is the probable yield loss for scenarios 2 and 2d, and the horizontal bar at 2000 is the probable yield loss for scenario 7.

An interesting relationship to the historic trends in agricultural yields over the past century is shown in Figure X-2, adapted from the work of Wittwer (1979). Agricultural yields were relatively unchanged until just before World War II, when a sharp increase began, continuing until 1968. At that time a "plateau effect" occurred, in which yields no longer increase as much in relation to the continuing technological inputs and agricultural practice improvements. Many explanations are possible as to the origin of this plateau effect, and Wittwer suggests ten possibilities. Here we need to consider the possibility that the response is an effect traceable, in part, to the emissions from fossil fuel combustion. Although other factors may have contributed, the widespread crop reductions due to oxidants, a phenomenon principally of the past decade (Figure X-1), should be considered as contributing to the plateau effect.

To determine the long-term implications of oxidants for CO_2 exchanges, one needs to evaluate whether these ozone levels will affect major carbon pools, such as forests. Evidence of ozone-induced foliage injury, reduced growth and mortality of selected forest species is available for California as well as the eastern United States (see Section IX). The forest decline in California's San Bernardino National Forest represents a major step toward emissions-induced species conversions--in this case toward chapparal species of lower productivity. If present trends toward lower forest productivity hold (Section IX), the forest losses could lead to a "diminished CO_2 sequestering capacity" in the temperate zone forests of up to 1×10^9 tons of carbon per year, almost 20% as much as the total fossil energy combusted in 1979. Other interactions between biospheric carbon exchanges and the emissions from fossil fuel combustion include the increased availability of alternate electron acceptors which are likely to increase the release of CO_2 from lake sediments by an amount unknown at this time. The significance of present and future SO_x emissions for effects associated with acidic deposition may be very important, however, for the reasons outlined in the next section.

AN AGGREGATE ACID DEPOSITION RESPONSE

Limited Objectives for an Interim Acid Rain Assessment

Although several large-scale studies are underway to determine more precisely the magnitude of the presently observable effects from acid deposition in eastern North America, these results were not available during the period of this ecological effects support study. Several preliminary assessments of acid deposition rates and related processes were completed during the past two years, however, to assess the consequences for acidic rain due to energy development scenarios in the Ohio River Basin. However, one must not only review the effects properly attributable to acid precursors, but also try to determine what part of the acid deposition is attributable to the ORBES region emissions. One must also ask whether the answers concern probable responses over the period of facility siting and construction (from now to the year 2000), or responses during the life of facilities constructed by 2000 (i.e., until the year 2030). The following paragraphs seek only to pose these ques-

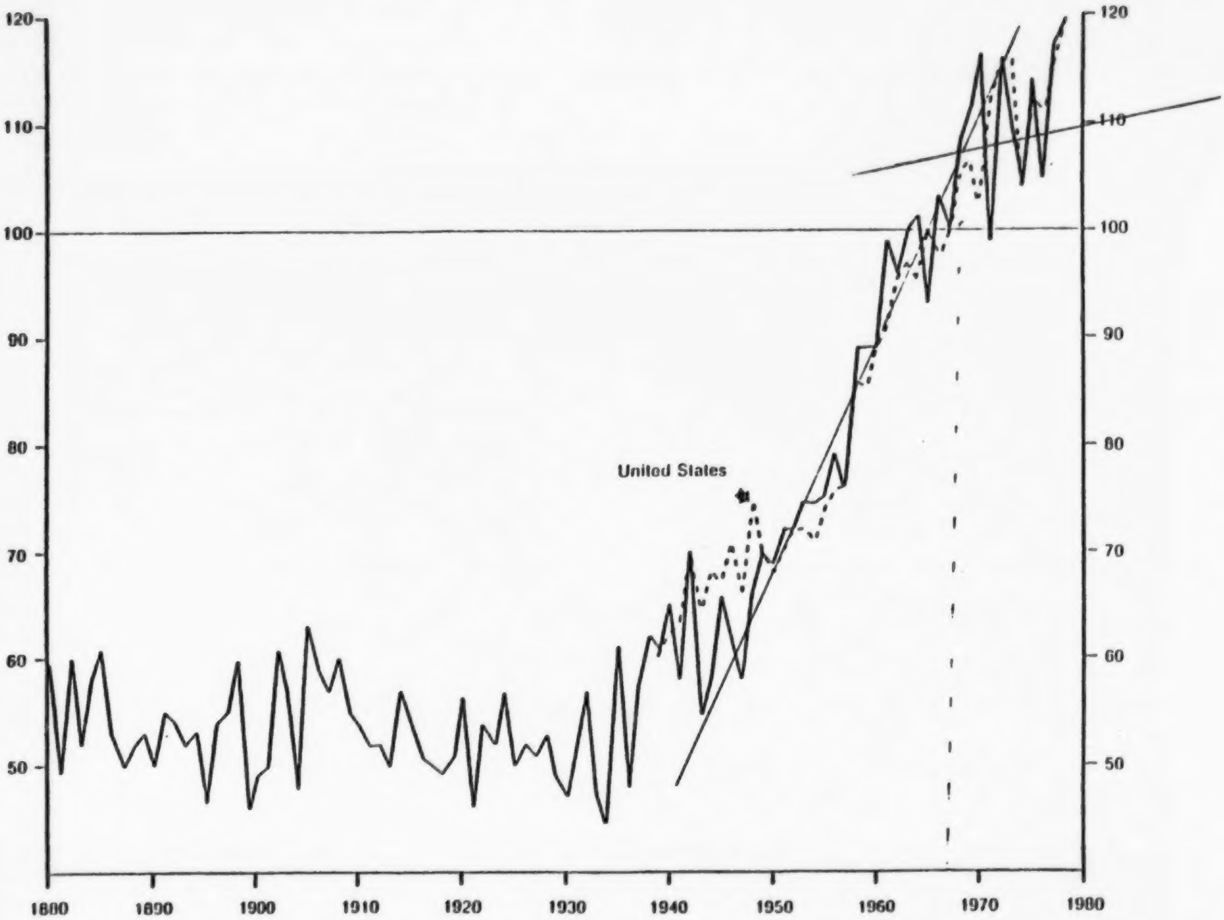


Figure X-2. Historic trends in agricultural crop yields per hectare in Michigan between 1880 and 1978. Dotted line represents total U.S. agricultural productivity (1967 = 100). Adapted from Wittwer 1979.

tions more precisely in relation to what is known about ORBES emissions projections.

The Phenomenon of Acidic Deposition

Although questions remain about some details of atmospheric chemistry, acidic substances are known to be produced in the atmosphere when SO_2 and NO_x from fossil fuel combustion combine with oxygen to produce sulfuric and nitric acids, as well as dry particulate sulfate and nitrate compounds. These pollutants can be carried hundreds of miles from their source. Some of the acid pollutants drop out as particles--a process called dry deposition--but more are washed from the air by rain, snow, or mist, yielding acid rain. The result is the acidification of surface waters in areas of sensitive soils, such as the Adirondack Mountain lakes and streams.

The dissolution of these pollutants in water in the atmosphere can increase the acidity of rain 100 to 1,000 times--from a pH of 5.6 for "normal" rain to a pH of 3.0, now being observed occasionally in parts of New England. Vinegar, at a pH of 2.6, is a key reference point. Acidity is measured by a logarithmic scale of ionic dissociation that ranges from pH 0 (very acid) to pH 14 (very alkaline). Each unit drop in pH represents a ten-fold increase in acidity. Distilled water has a pH of 7; "normal" rain is slightly acidic due to the dissolved CO_2.

In addition to acid in rainfall, acidic substances are produced by dry deposition of SO_2 and NO_x as gases absorbed by vegetation and soil. During the growing season, relatively large quantities of SO_2 and NO_x gases are taken up by plant foliage. After death and normal decomposition of the organic matter so formed, these sulfur and nitrogen compounds contribute significantly to the acidification of natural resources (Cowling 1980).

But many ask, "What is the source of the acidic deposition on a remote Adirondack lake?" Consider the gaseous emissions from any large urban or industrial area. Emissions build up to moderately high values and very reactive concentrations, and then move out over the rural landscape. Many questions remain as to distance transported and ultimate effects. However, the technique of air parcel trajectory analysis can back-track the path of an air parcel to its original location 12, 24, and 36 hours before it deposits acid rain. Miller, Galloway and Likens (1978) found that trajectory analysis, supplemented by meteorological maps, identified the high industrial, coal-burning regions as source areas of the constituents in acid precipitation. The technique is becoming increasingly precise, and analysis of source areas requiring abatement will be a part of acid deposition management.

Recent reviews have shown that, despite some questions of rates of reactions and control mechanisms, we know enough about the acidic deposition phenomenon now to describe the general relationships, the areas of impact, the mechanisms of effect, and the basis for some lags in response times. Figure X-3 shows the principal ecological consequences resulting from formation and deposition of elevated hydrogen ion and sulfate anion densities (sulfate and nitrate), the consequence of atmospheric chemical reactions following release of nitrogen and sulfur oxide emissions from utilities, transportation, and other industries. The ultimate effect of this deposition must be viewed as a coupled system which, for simplicity, is shown here as a flow diagram. The principal findings of Cowling (1980) in the area of emissions, transport, and deposition are shown in Table X-3.

Treating the acid precipitation phenomena as a linked system (Figure X-3) leads to considering effects of elevated hydrogen ion density (pH) as the consequence of inputs from anthropogenic sources. The greater part of these inputs are believed to derive from fossil fuel combustion emissions by elevated stacks (Miller et al. 1978). These emissions remain in the lower atmosphere much longer than short-stack emissions, long enough for acidifying reactions to be complete.

Thus, the following analysis is developed around the known and projected data on fossil fuel emissions, transport and transformation of these products to form hydrogen ion (H+) and related anions in precipitation; the present data on known effects; projections of trends in acid rain effects based on projected emissions; and the nature of responses and lag-times in soils, watersheds, streams, and lakes.

Present and Projected Combustion Emissions

Projections of the magnitude of acidic deposition in the future, and therefore of the implications for natural resources, must begin with an inventory of present and projected emissions. The projections for change in acid-forming precursors from utility development in the Ohio River Basin are shown in Figure X-4. A potential 40% reduction in NO_x plus SO_x appears possible under some scenarios by the year 2000, but other scenarios indicate very little reduction is possible. These estimates do not reflect the recently proposed increases in SO_2 sources during the decade of the 1990's. Automobile sources of acid deposition seem likely to move downward by the year 2000, but long-distance transport of emission products from coal development in the central and southern plains states is likely to add to the problem in the northeast. Thus, depending on decisions now being made, the total precursors of acidity in precipitation could increase significantly, might increase only slightly, or could, with rapid conversion to solar energy sources, decrease as much as 30% between now and the year 2000.

Data Based on Recent H+ Inputs

Evidence is available as to the long-term trends in H+ deposition from rainfall and snowfall. The first general data base for the pH of rainfall (from Cogbill and Likens 1974) covered the 1955-56 period (Figure X-5). The second data point is available for the 1972-73 period (from the same authors). More recent data are available from the newly established National Atmospheric Deposition Program. The entire sequence shows a

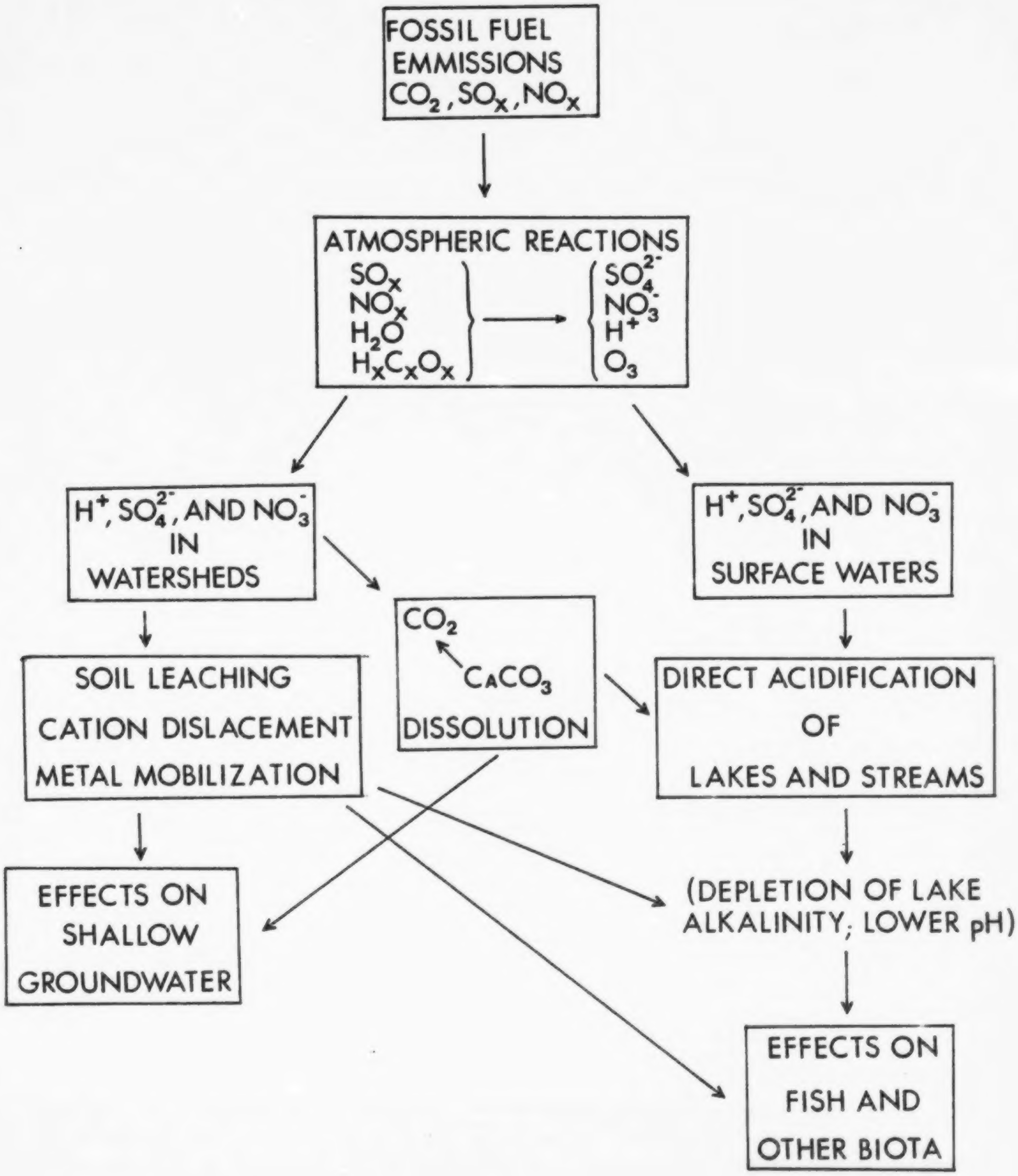

FOSSIL FUEL EMMISSIONS CO_2, SO_x, NO_x
ATMOSPHERIC REACTIONS
SO_x NO_x H_2O $H_xC_xO_x$ → SO_4^{2-} NO_3^- H^+ O_3
H^+, SO_4^{2-}, AND NO_3^- IN WATERSHEDS
H^+, SO_4^{2-}, AND NO_3^- IN SURFACE WATERS
CO_2 ← $CaCO_3$ DISSOLUTION
SOIL LEACHING
CATION DISLACEMENT
METAL MOBILIZATION
DIRECT ACIDIFICATION OF LAKES AND STREAMS
(DEPLETION OF LAKE ALKALINITY, LOWER pH)
EFFECTS ON SHALLOW GROUNDWATER
EFFECTS ON FISH AND OTHER BIOTA

Figure X-3. Flow diagram showing system linkages for acid precipitation formation, deposition, and effects as a consequence of nitrogen and sulfur oxide emissions from fossil-fuel combustion.

TABLE X-3. A CONSENSUS ON PRESENT SCIENTIFIC UNDERSTANDING OF ATMOSPHERIC ACID FORMATION, SOURCES, AND DEPOSITION. (After Cowling 1980)

Property/ Constituent	Responses
1. Geographic area	The eastern United States, Canada, Europe, and Japan, and locally in the western United States.
2. Sources	From elevated total emissions of SO_2 and NO_x, transformed in the atmosphere to sulfates, nitrates, and sulfuric and nitric acids. Contemporary acidification results from both natural (5% to 10%) and man-made emissions (90% to 95%) of gases, aerosols, and particulate matter.
3. Emission height	Tall stacks decrease ground-level concentrations of SO_2 and NO_x but increase the geographical area of acid deposition.
4. Transport	Air-mass movement and chemical transformation studies indicate acid precipitation in one state or region of the United States or Canada results in large part from emissions in other regions, up to hundreds of miles from the original source.

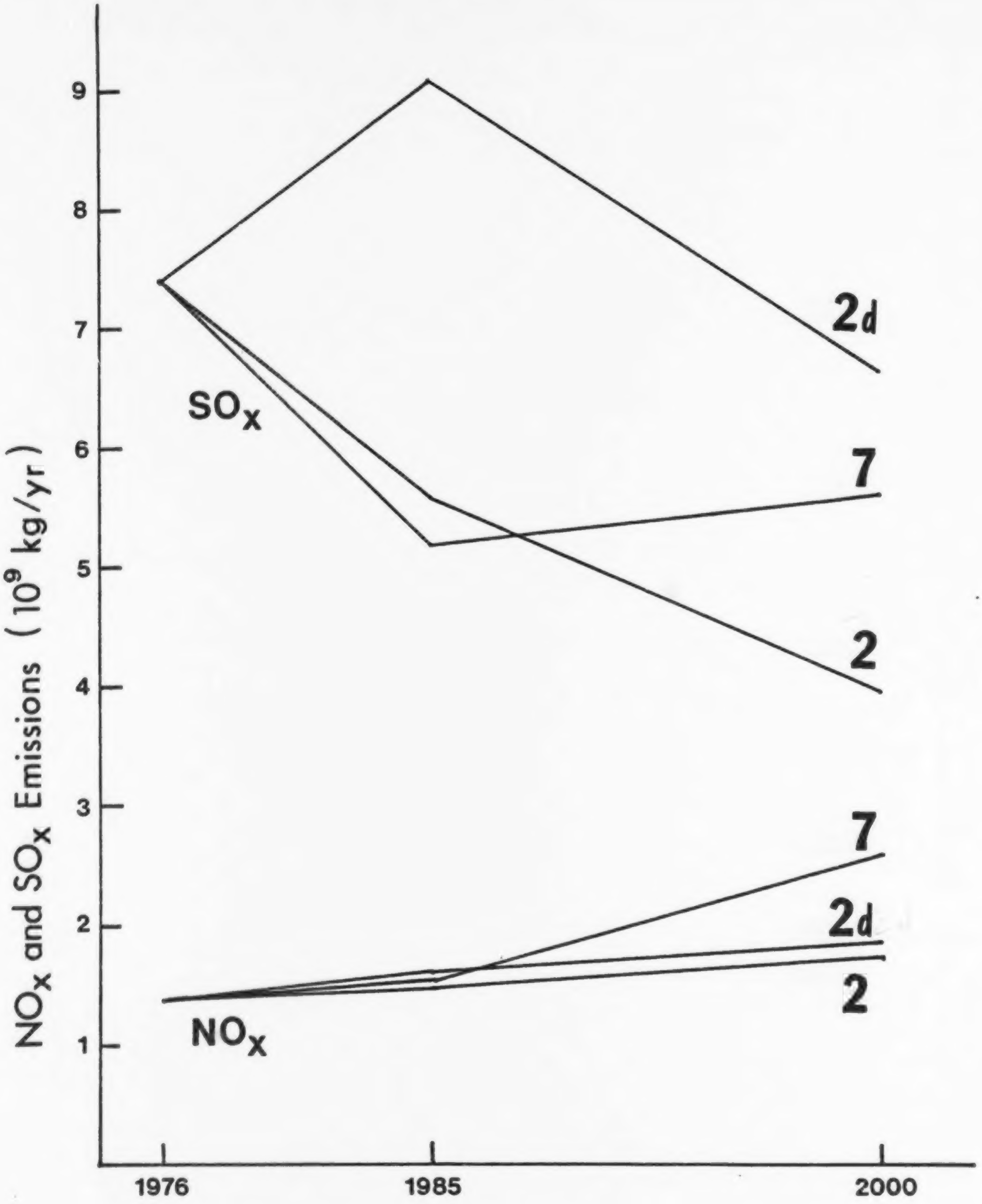


Figure X-4. Projections of NO$_x$ and SO$_x$ emissions (1980-2000) under three scenarios of utility development in the Ohio River Basin.

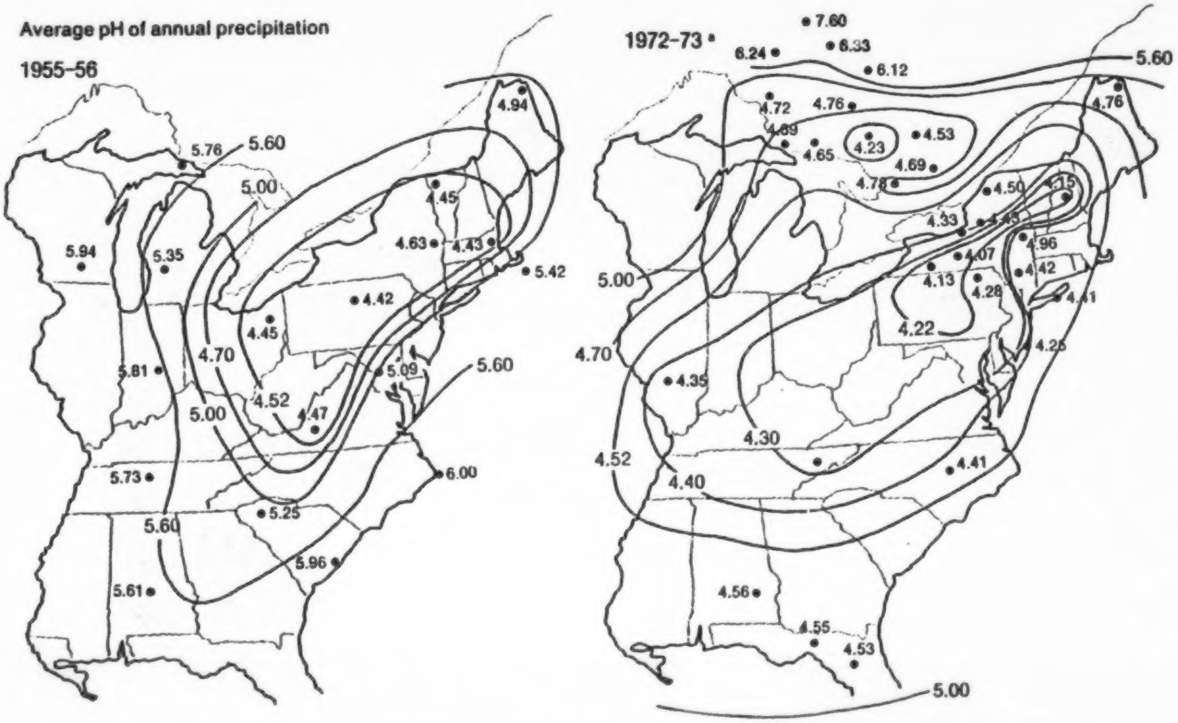

Average pH of annual precipitation
1955–56
5.76
5.60
5.00
4.94
4.45
4.63
4.43
5.42
5.94
5.35
4.42
4.45
4.70
5.81
5.09
5.60
4.52
4.47
5.00
5.73
6.00
5.25
5.60
5.96
5.61
1972–73
7.60
6.24
6.33
6.12
5.60
4.72
4.76
4.69
4.65
4.23
4.53
4.69
4.78
4.76
4.50
4.15
4.33
4.45
4.96
4.07
4.13
4.42
4.28
4.41
5.00
4.22
4.70
4.35
4.25
4.30
4.52
4.40
4.41
4.56
4.55
4.53
5.00

Figure X-5. Precipitation pH isopleths from 1955-1956 to 1972-1973 showing the time-course of change in H^+ loading for the northeastern United States (Cogbill and Likens 1974).

clear increase in the intensity of H+ deposition (i.e., depression of pH) from 1955 to 1976, and a very large spread in the geographic area over which H+ density in precipitation is elevated. Recent results show a spread to western Wisconsin and Minnesota and significant decreases in precipitation pH in the Rocky Mountains and Sierra Nevada range of California, both areas with locally sensitive bedrock and soils (Lewis and Grant 1980, McColl 1980).

Consider now the prospective trends in deposition if we accept the three broad emission scenarios (Figure X-4) for the Ohio River Basin. These can be summarized in terms of total annual deposition of sulfate and nitrate anions. Data on total sulfate and nitrate deposition at the Hubbard Brook watershed in New Hampshire are summarized in Figure X-6 (from Likens *et al*. 1977). The historical background estimate is from Glass and Loucks (1979) for a northern Minnesota site representative of northern continental (i.e., non-coastal) locations. An estimate of 5 to 10 % natural background in the relatively polluted continental regions is now widely accepted in Europe as well as North America. The dashed lines show first a linear projection of continued increases in anion deposition, and, alternatively, a trend toward leveling off of the deposition rate (and therefore of H+ input) by the year 2000. For the reasons considered above, however, a sharp decrease in deposition in H+ or the anions, although possible, must be regarded as improbable.

Inventory of Present Effects on Natural Resources

The effects of acidic precipitation on plants are only one facet of complex atmospheric/plant/soil interactions. Both injurious air pollutants and required plant nutrients are readily absorbed through foliage from the atmosphere, as well as through the soil solution. The growth and productivity of plants are governed by the availability and balance among fifteen elements essential for growth. Some of the elements, although essential, can be injurious to plants at abnormally elevated levels. For example, small amounts of sulfur and larger amounts of nitrogen are needed for the synthesis of protein in plants, but excess amounts of gaseous oxides of sulfur and nitrogen, and sulfuric and nitric acid aerosols, are injurious to plants at very low concentrations. In general, however, the above-ground portions of plants have rather thick protective surfaces, and few serious direct effects are observed, as summarized in Table X-4.

For soils and watersheds, the potential effects of acid rain have been described by Norton (1976) as trends toward solution of clay materials, loss of nutrients and exchange capacity, increased rates of chemical denudation, mobilization of metals, and expansion of podzolization. Iron and aluminum oxides and hydroxides can be dissolved and go into soil solution, where they are toxic to some plants and animals. A number of investigators have discussed possible implications of acid rain to agricultural soils. Considerable acidification has occurred here over the past 25-50 years as a result of current fertilization practices with ammonium sulfate and ammonium nitrates, which tend to acidify soils. However, for agricultural soils, atmospheric acid deposition probably is

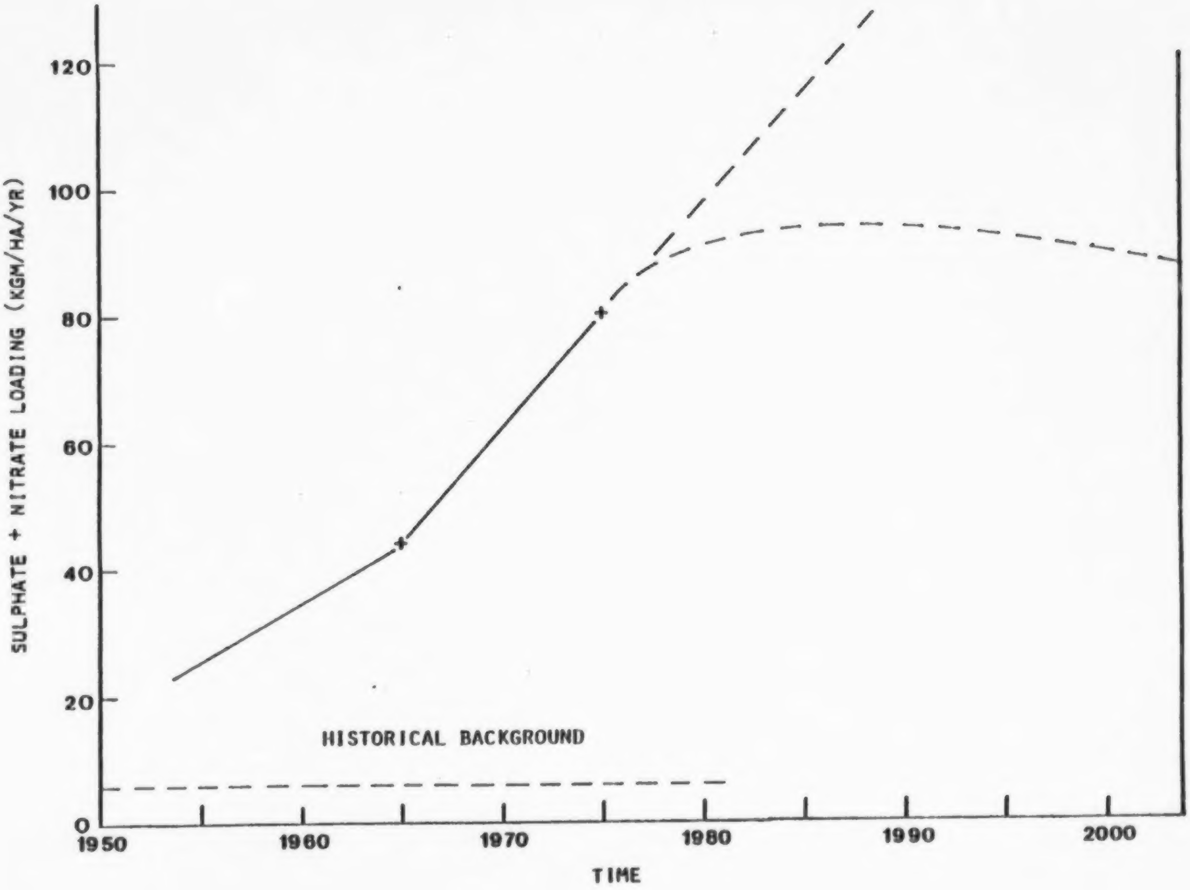


Figure X-6. Historical background (ca. 1850), recent observations (1965-1977), and projections (1980-2000) of total annual deposition of sulfate plus nitrate. Recent data are from the Hubbard Brook watershed in New Hampshire (Likens et al. 1976); the historical background estimate is for northern Minnesota (Glass and Loucks 1979).

Table X-4. Consensus on present scientific understanding of atmospheric acid effects on plants. (After Cowling 1980.)

	Property/ Constituent	Response
1.	Potential injurious substances	Includes not only acidic substances, but also elements such as Mn, Zn, Cu, Fe, Mo, B, F, Br, Al, Pb, I, Ni, Cd, V, Hg, As.
2.	Essential nutrients	Mn, Zn, Cu, Fe, Mo, and B are needed by plants in small amounts. At higher concentrations, however, each of these (as well as S and N) can be toxic to plants and animals; the acidity of precipitation affects the solubility, mobility, and toxicity of certain of these elements.
3.	Damage to plants	Most likely when a particularly sensitive life-stage, developing in poorly buffered waters or soils, coincides (in time and/or space) with major episodic acid inputs or other injurious substances.
4.	Crop and forest	Direct and indirect damage reported in laboratory, greenhouse, and field experiments using synthetic rain. The biological effects include: - Induction of necrotic lesions on foliage - Loss of nutrients due to leaching from leaves - Predisposition of plants to infection by pathogens - Accelerated erosion of waxes on leaf surfaces - Reduced yield of marketable crops - Inhibition of nodulation of legumes leading to decreased nitrogen fixation and - Reduced rates of decomposition of leaf litter and decreased mineralization of organically-bound nutrients
5.	Economic significance	Reliable evidence of economic damage to agricultural crops and forests, and to biological processes in soil, by naturally occurring precipitation and dry deposition of aerosols has been reported only rarely.

overwhelmed by the cation exchange capacity inherent in good agricultural soil management. Still, serious effects of acid precipitation are known for shallow, poorly-buffered (i.e., low in calcium) natural soils (mostly in rocky or mountainous recreational areas and wilderness). In general, the results are as follows:

- lowered soil pH
- accelerated leaching of plant nutrients and other ions
- accelerated weathering of soil minerals
- changes in soil biota
- reduction in organic matter decay rates and associated release of plant nutrients
- reduction in nitrification
- reduced phosphorus availability to plants
- increased aluminum mobility and associated toxicity
- increased mobility of organic soil components
- changes in anion-cation balances necessary for plant growth

The earliest identified responses of aquatic resources to acid deposition in North America are those described by Beamish and Harvey (1972) from studies in 1969-70 in the LaCloche Lakes north of Georgian Bay, Ontario. Here, by 1970, an unusually large number of lakes had pH values as low as 4.5, and a large number showed complete loss of fish populations at that time. Reproductive failure begins for some species at lake pH values of 5.5, and is serious when a lake drops below pH 5.0. The most serious drop in pH occurs as the winter accumulation of H+ is stripped from snow during snowmelt. Not all of the lakes were reduced in pH to a biologically significant level in one year, however, indicating the responses observed in 1979 were the accumulation of an annual "addition" of lakes into an "affected state," as determined by the continuing addition of acidity. The rate of this annual addition of affected lakes is still a major unanswered question, but for the LaCloche Lakes area, it appears to have been 5 to 10 additional lakes acidified annually during the late 1960's.

Later, study of lake chemistry and fish populations in high-elevation lakes of the Adirondack region of New York showed a marked decline in the pH of the lakes compared with 40 years earlier (Figure X-7). The decline in pH here also was associated with failure of fish to reproduce and maintain populations in the youngest age classes. Although some questions about response rate and time lags remain, the effects summarized here must be regarded as a cumulative phenomenon, in which some lakes annually reached a defined H+ density threshold (e.g., in the order of pH 5.0) at which significant effects on fish reproduction can be expected. These effects may be in calcium metabolism, sodium and potassium balance, or

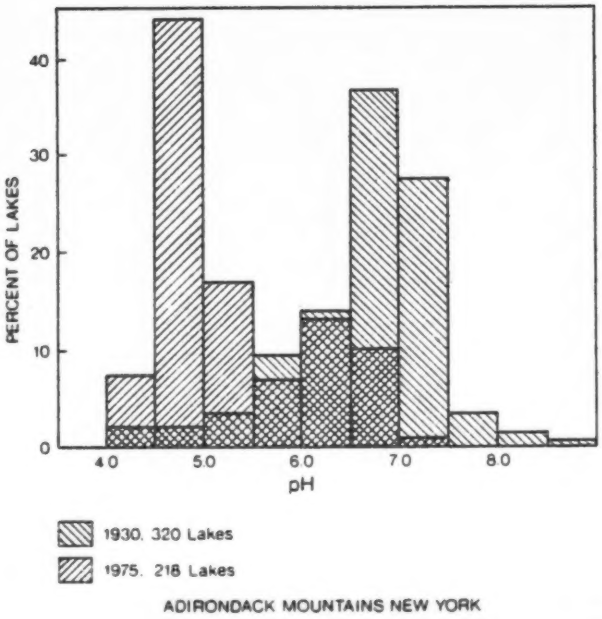


Figure X-7. Frequency distribution of pH in lakes in the Adirondack Mountains, New York, in the 1930's. (Redrawn from Schofield 1976)

direct toxicity from metal ions such as free aluminum (Al^{3+}) mobilized into water by excess H^+. Of the 218 lakes measured in 1975, approximately 50% are significantly acidified, having passed this threshold within the previous 10 to 15 years--an annual rate of 7 to 10 lakes per year. These will be aggregated into annual rates for larger areas in a later section. Monitoring of lakes in Ontario, Quebec, Nova Scotia, and throughout New England shows large numbers of lakes with depressed pH levels (Harvey, personal communication). Many of them are approaching the point at which fish populations will be affected. However, detailed data on biological effects in these systems are not available at this time. Table X-5 provides a summary of the known effects (from Cowling 1980).

A special case is the rapid decline in pH and collapse of the salmon fishery in Nova Scotia rivers. Representative here are the pH measurements in the Tusket River of Nova Scotia between 1955 and 1975 (Figure X-8, from Thompson _et al_. 1980). The salmon fishery, which for over 200 years had survived so many other pressures in at least modest strength, has now disappeared entirely in all seven rivers. The mechanisms for this collapse, while not yet known in all details, are almost certainly due to interactions between elevated Al^{3+} in the low pH water and gill function, particularly oxygen uptake and ionic regulation of the bloodstream.

This lake and stream response information, taken in conjunction with earlier records from northern Europe, indicates a trend in the number of lakes affected similar to the upward trend seen earlier in total H^+ loading during the 1950's and 60's (Figure X-9). The studies by Dickson (1978) show that the pH of lakes in extremely sensitive surroundings drops sharply as the annual sulfate loading reaches 15 kilograms/hectare/year (kg/ha/yr). Lakes in slightly less sensitive surroundings become increasingly acid as deposition increases from 30 to 60 kg/ha/yr. These effects, taken in conjunction with the evidence for annual increases in the number of lakes reaching a pH of 5.0 or less, suggest the depression in lake pH values could expand now to larger watersheds and larger numbers of lakes in the years ahead, even with no increase in acid precursor emissions. This is not a conclusion at this time, but a hypothesis to be evaluated. If we follow the scenario projecting continued increase in acid-forming emissions, we do know that much larger geographic areas will be affected by the elevated levels of acid deposition.

Trends for Future Lake and Stream Responses

The first step in exploring natural resource implications is to examine prospects for continuation, expansion, or abatement of acid rain threat. These scientific questions and the effects are remarkably similar to the mechanisms of DDT-induced effects on resources, summarized qualitatively by Rachel Carson in 1962 and documented in detail in the decade which followed. We know now for acid rain, as many scientists knew in 1962 for DDT, what the sources, mechanisms, and future responses are likely to be; but there are now, as than, certain differences in scientific judgments that prevent effective, but costly, control measures from being implemented. Thus, in the case of acid rain effects, we must explore as best we can what are the risks of postponing ameliorative action.

Table X-5. Consensus on present scientific understanding of atmospheric acid effects on aquatic organisms. (After Cowling 1980.)

	Property/ Constituent	Response
1.	Reproduction	Inhibited for many species of aquatic organisms at pH's between 6.0 and 5.0.
2.	Adult mortality	Populations of many freshwater fish go to extinction at a pH below 5.0; aluminum toxicity is probably partly or largely responsible at these pH's.
3.	Status of fishery	As of 1979, several hundred lakes in the Adirondack region of New York, and many hundreds of lakes in southern Ontario, Quebec, and Nova Scotia, showed acid stress in the form of diminished populations or extinction of fish populations.
4.	Toxic ion effects	Acidity increases the mobility of many toxic cations, including Al, Mn, and Zn, increasing their concentrations in adjacent streams and lakes. These toxic ions are transferred from soils into surface and ground waters.
5.	Ecosystem linkages	Neighboring terrestrial and aquatic ecosystems usually are so intimately linked that evaluation of effects in the aquatic system without considering effects in the terrestrial system is unrealistic.

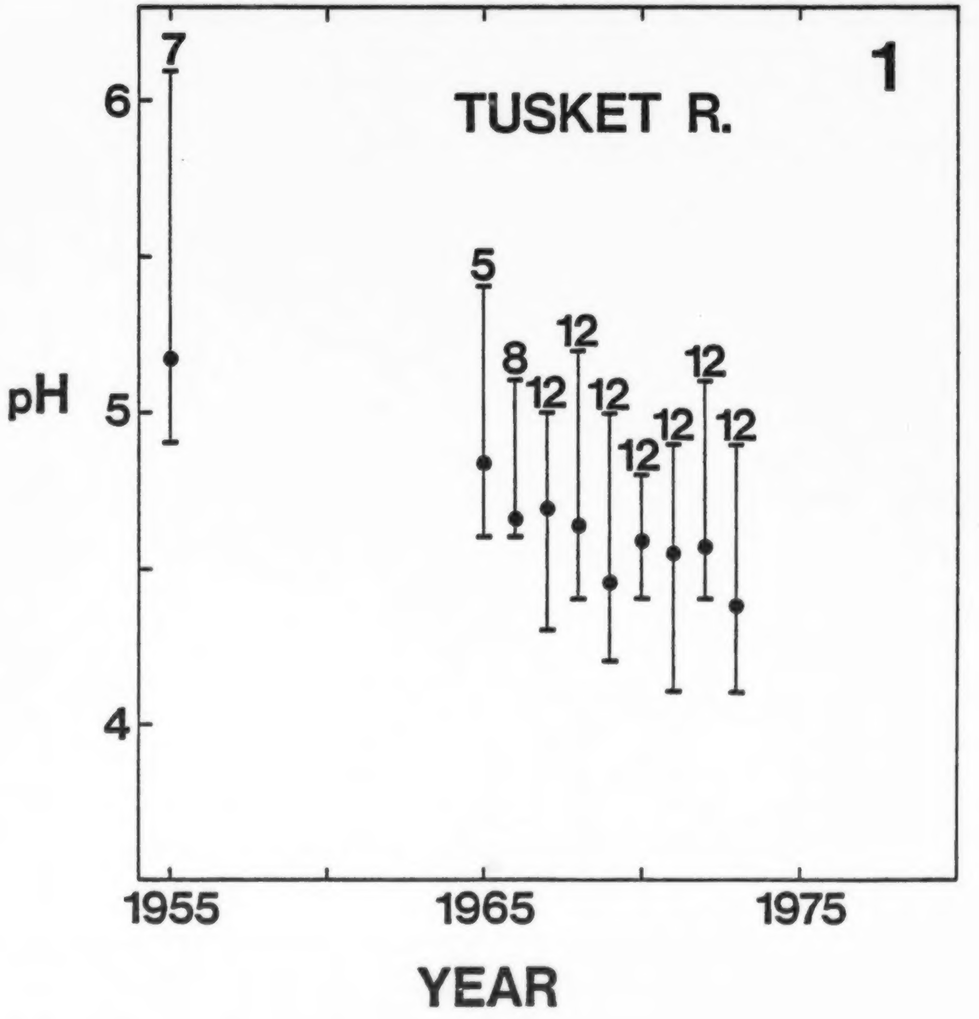


Figure X-8. Measurements of pH in waters of the Tusket River of western Nova Scotia between 1955 and 1975. (From Thompson et al. 1980)

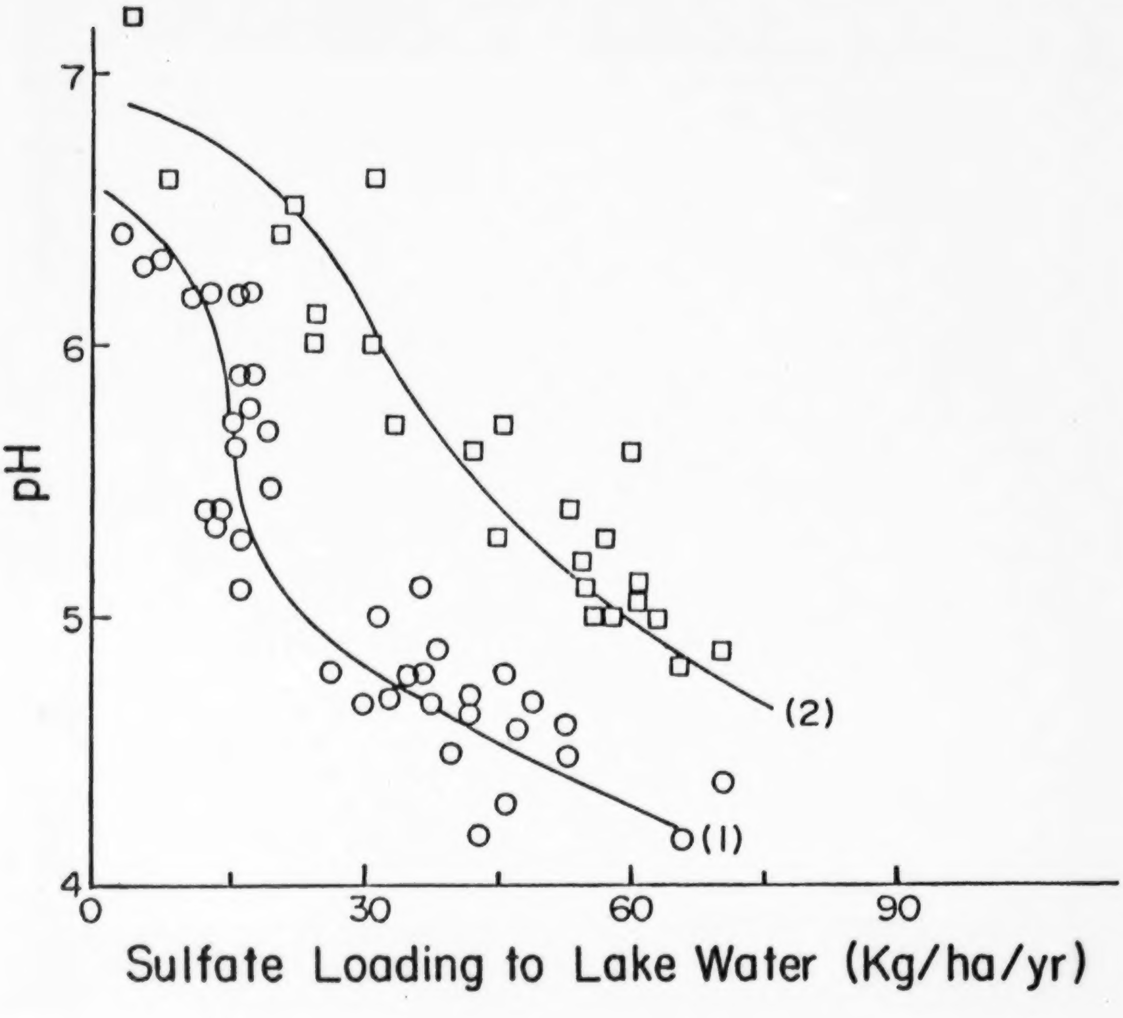


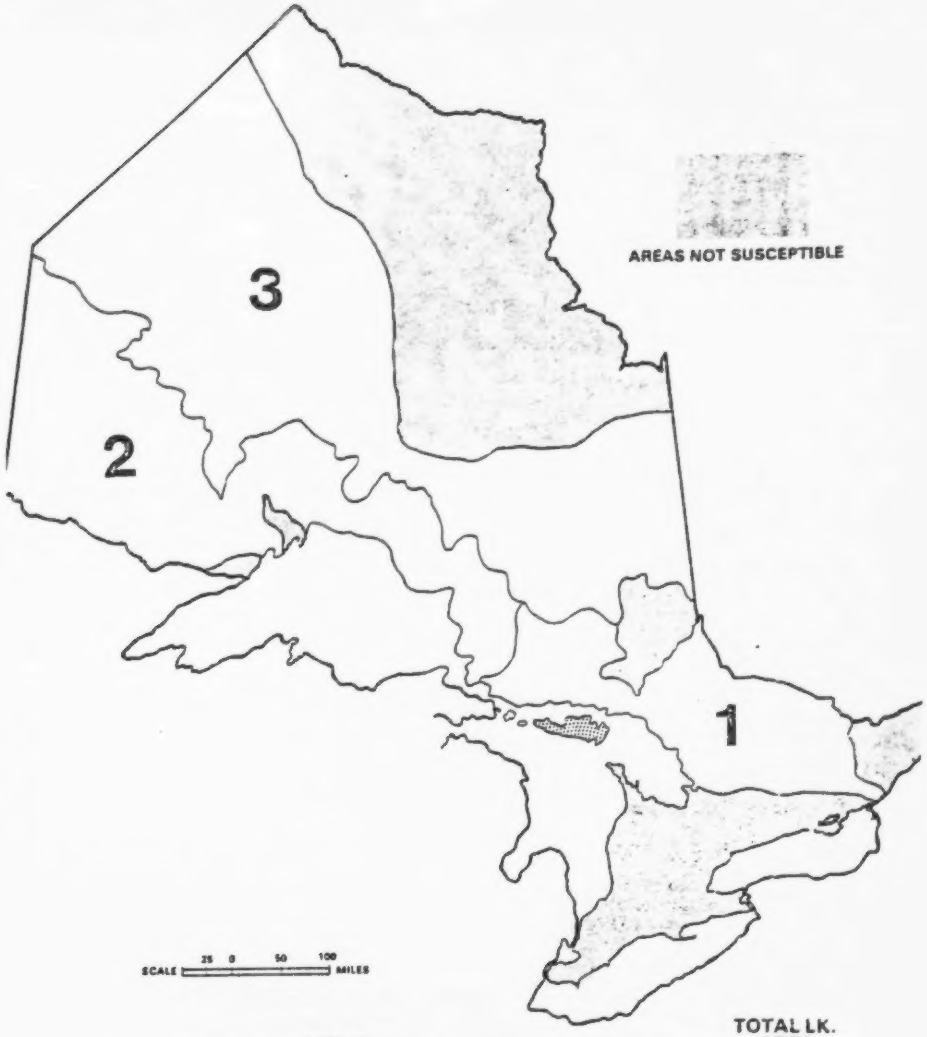
Figure X-9. Relationship between acid loading and pH change for lakes in Sweden in very sensitive (1) and somewhat less sensitive (2) surroundings. (Dickson 1978)

What do the current studies indicate? A preliminary estimate of the total number of lakes that may experience acidification by the year 2000 has been prepared by the Ontario government (Figure X-10, Province of Ontario 1979). This estimate suggests some 48,000 lakes in Ontario will experience significant acidification within the next 20 years. An inventory of present alkalinities of lakes on the Canadian Shield in Ontario, the levels of acid inputs currently known for the area, and the general relationships for acidification were used in making this estimate. Since many aspects of the lake response to acid inputs are still unknown, this estimate does not necessarily mean all of these lakes will experience a significant failure of fish reproduction; but effects on some species are almost implicit once significant acidification has begun. The Ontario estimate suggests a total of 3,000 lakes per year (upper curve, Figure X-11) will reach a significant, defined threshold of acidity during the two decades between now and the year 2000.

A more conservative view of the relationships between inputs and probable responses in lakes is also suggested in Figure X-11. Two different deposition response scenarios are shown, each associated with certain assumptions as to acid precursor emissions. Considering the emissions first, the probable maximum abatement of acid precursors shows an approach toward 20% of present levels by the year 2030, probably the most optimistic possible. We cannot truly estimate the recovery in lakes by this abatement. A linear upward trend in emissions, and therefore of acid loadings, is also possible for the reasons discussed previously.

Figure X-11 also suggests a way to express the annual effect on lakes as a rate at which additional lakes and streams experience a depression of pH (below 5.0) and, therefore, associated effects on fish populations. Extrapolating from the small area sampled by Beamish in 1970 (Beamish and Harvey 1972) to other regions of similar bedrock and similar acid deposition rate at the time, one obtains a preliminary estimate of 50 lakes annually reaching a pH threshold significant for fish reproduction throughout Ontario, Quebec, New York, New England, and the Maritime Provinces. The later estimate of 7 to 10 lakes per year within the high-elevation Adirondacks area (Schofield 1976; see the earlier text) can be extended to about 20 other similar areas in eastern Canada and northeastern United States, suggesting an addition of 200 lakes annually at that time.

Although these two estimates of annual addition to the inventory of affected lakes are approximate, the general upward trend during the 1970's is not in doubt. What does the future hold? Three projections are shown in Figure X-11, using the preliminary results just cited. The line of circles represents a linear projection of the annual rate of lake "losses" based on the Ontario estimate of 48,468 lakes by the year 2000, for Ontario alone. The lower dashed line represents a linear projection of the trend evident for lakes during the past decade in the northeastern part of the continent, allowing no time lag in relation to changes in inputs. The solid line is simply an intermediate, reflecting an assump-



CODE	TOTAL # OF LAKES	MEAN AREA/LK. (ACRES)	PERCENTAGE SUSCEPTIBLE	# OF LKS. SUSCEPTIBLE	TOTAL LK. AREA SUSCEPTIBLE (SQ. MI.)
1	40,589	58.9	50%	20,295	1,868
2	76,728	98.0	20%	15,346	2,350
3	64,133	130.0	20%	12,827	2,605
TOTALS:	181,450			48,468	6,823

Figure X-10. Estimate of total number of lakes in the Province of Ontario susceptible to acidification by the year 2000, based on present alkalinities, levels of currently-known acid inputs, and a generalized acidification relationship. (Prov. of Ontario 1979)

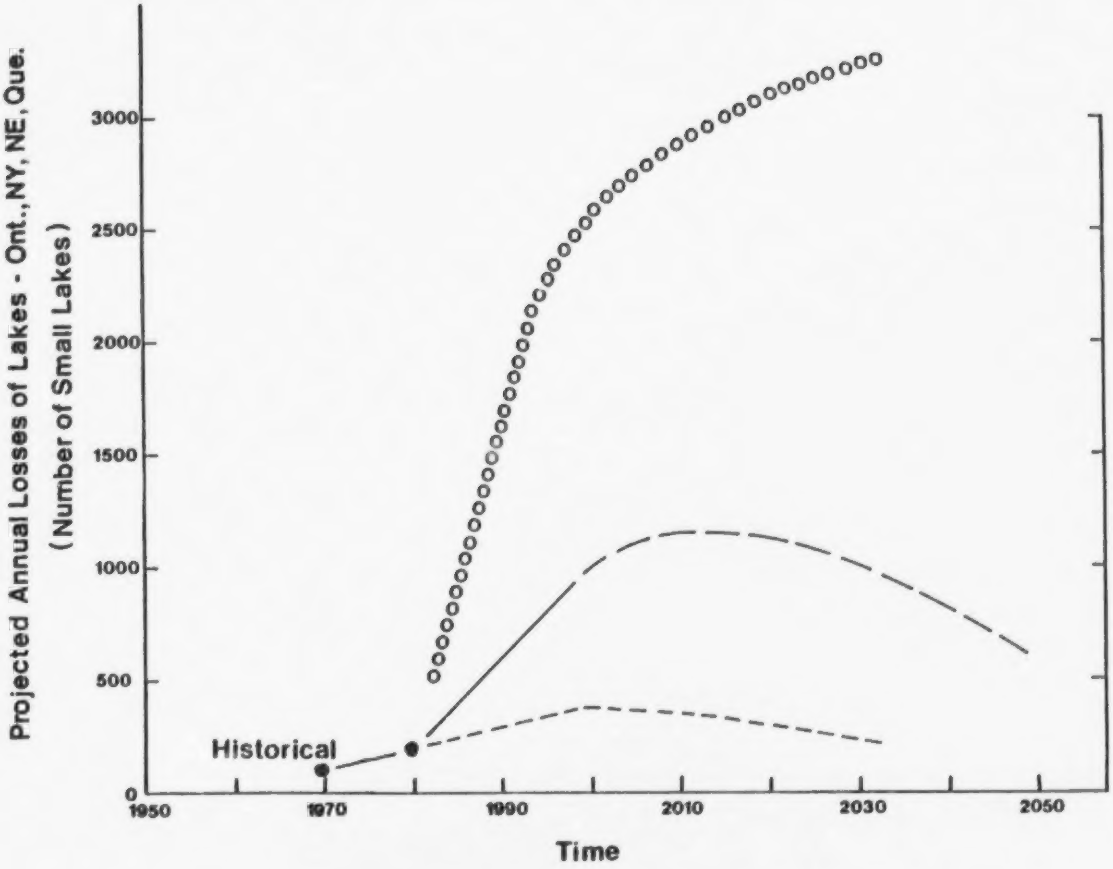


Figure X-11. Probable response times for lakes of Ontario, Quebec, New York, New England, and the Maritime Provinces subjected to acidic deposition. The circles represent an approximate projection of the annual rate of lake "losses" by an Ontario estimate; the lower dashed line represents a linear projection of the recent trend evident in the northeastern part of the continent, assuming no time-lags; and the solid line represents an intermediate projection which assumes some time-lag will result in higher rates of expression of acidification effects during the 1980's and 1990's.

tion that time lags following the high loading rates of the 1970's will be seen as higher rates of expression of lake acidification (in terms of number reaching a biologically significant threshold) during the 1980's and 90's.

Unfortunately, we do not have, in either Canada or the United States, any direct monitoring of the effects on aquatic resources. Thus, the trend projections shown here must be viewed as questions in need of answers.

Although we know much about the status of atmospheric acid inputs and of aquatic resources in relation to acidity of lakes and streams, little is known yet about the response of wildlife other than fish. Certainly the pH of spring pools appears to be too low for the reproduction of many amphibians, but salamanders are the only group in which effects are well documented. Scattered observations have been made of searches by osprey in lakes that have no fish; they persist in the search for a season and are then not seen. We do not know whether the pairs relocate. Since crayfish, snails, and other species with a high calcium demand are among the earliest to disappear in acidified lakes, wading species such as the blue heron are likely to be affected, although no records are being made that I have been able to discover.

More pathetic are the observed effects on nesting pairs of loons on lakes in Ontario that have no fish. Although the adults apparently survive the season, perhaps by using other lakes in the area, it appears the nestlings do not survive. No loons return in succeeding years (H. Harvey, personal communication).

Management options range from massive additions of lime to breeding of genetically resistant strains of fish, and ultimately, to abatement of the acid precursors. Treatment of the presently acidified waters would cost on the order of $250,000,000 annually. However, even if funding was possible, it is very difficult to reach all the remote streams and lakes during the critical snowmelt period. Also, many of the effects are the result of Al^{3+} liberated to groundwater from the soil, and lime additions in the lake do not control these stream and lake-edge inputs.

Development of resistant fish strains has been an attractive option for some resource managers. However, fish live in a complex food-web, within which many other species, particularly crayfish and some mayfly species, also are sensitive to acid inputs to the lake or stream. It does not appear possible to breed resistance for the entire food chain, so such a program has serious limitations. Control of fossil energy emissions appears likely to be the most feasible and cost-effective option.

CONCLUSIONS

The contribution to natural resource systems in eastern Canada and the northeastern U.S. from the acidic deposition apparently originating with ORBES region utility emissions is more difficult to estimate than the contribution of ORBES air pollutants to agricultural and forest losses. In one sense, the total SO_2 and NO_x sources from the region (utility as well as mobile and other industry sources) can be added to estimate partial totals of precursors contributed from remote sources to the various "deposition regions." However, there also is partial local scavenging of ORBES precursors before the transformation products reach sensitive regions, a part of the process producing the crop effects.

Thus, the total contribution of ORBES region emissions to acid-rain induced aquatic effects described above probably is best estimated (for the present) simply by its proportion of total emissions within a 500-to-700-mile radius "source area" contributing to deposition in the sensitive regions downwind. For the Adirondack and Ontario areas, estimates by McBean and Galloway* suggest the ORBES region proportion of precursors will be between 20 and 30 percent. A smaller percentage applies to the acidified lakes of Georgian Bay and New England. Within these estimates of 10 to 25%, the differences introduced by the three ORBES development scenarios described earlier (in which increases in NO_x emissions trade off to some extent against decreases in SO_2 emissions) would not be distinguishable.

Numerous sources of potential error could affect these projections, and acid rain research programs now underway may soon reduce the limitations that must soon be attached to relationships suggested here. Although the results could be expressed in terms of probable upper bounds and lower bounds, there is some likelihood that the distribution within these bounds has been reinforced by the effort to take a "most probable" position in each successive step. Assuming that the current inventory of lakes in Ontario represents about 40% of the lakes in the region of interest (excluding Michigan, Wisconsin, and Minnesota), an annual loss of 500 lakes represents a rate of 0.25% per year. It is also possible that H+ effects on alkalinity sources in watersheds will not be cumulative, and that some sources of buffering, other than metal mobilization, may emerge to prevent the continued depletion of cations to water systems.

Studies of lakeshore and streamside property values affected by a failure of the fishery have only begun. Losses due to decreased value of the fishery itself appear to be small, but annual loss of property values may well be large. Such capital losses also lead to inter-generational options lost, a consideration of some monetary significance.

These preliminary results suggest the total annual resource benefits likely to acrue from abatement of ORBES region utility emissions will be comparable in scale to the agricultural benefits estimated in previous sections. The total area of sensitive recreational resources where ORBES region precursors represent more than 5 to 10% of the acid deposition is about double the total agricultural acreage considered in the crop evaluation. Utility emissions, however, represent probably only half of the acid-forming precursors entering into long-distance transport and deposition.

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* Interim report of the U.S./Canada Research Consultation Group on Long Range Transport of Air Pollutants. Dept. of Environment Canada and U.S. EPA.

These lines of reasoning suggest that a "common currency" expression of the combined effects of acidic deposition, gaseous pollutant emissions and oxidants formed from ORBES region utilities would be about twice that from crop losses. More thorough analyses of these relationships are urgently needed, however.

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